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Radionuclide Migration Pathways Analysis for the Oak Ridge Central Waste Disposal Facility on the West Chestnut Ridge Site

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Energy Divison

RADIONUCLIDE MIGRATION PATHWAYS ANALYSIS FOR THE OAK RIDGE CENTRAL WASTE DISPOSAL FACILITY ON THE WEST CHESTNUT RIDGE SITE

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SUMMARY

A 506-ha (1,250-acre) tract of land on the Oak Ridge Reservation (ORR), referred to as the West Chestnut Ridge Site, has been proposed for the Central Waste Disposal Facility (CWDF). The site is bounded by Bear Creek Road to the north, Tennessee Highway 95 to the east, and the New Zion Patrol Road to the south and west. The CWDF will serve as a repository for solid, low-level radioactive waste from the Oak Ridge National Laboratory, the Y-12 Plant, and the Oak Ridge Gaseous Diffusion Plant. A dose-to-man pathways analysis for waste anticipated from the three facilities is made on the basis of the site characteristics, and the results are compared with the maximum permissible doses an individual may receive from a radioactive waste disposal site. The capacity of the site for the waste is determined and a buffer zone--beyond which human activity would be unrestricted--is recommended on the basis of the predicted zone of groundwater contamination.

The proposed design of the CWDF consists of shallow trenches with features that minimize both the contact and residence time of infiltrating water with the waste. For this disposal method, the predominant radionuclide migration pathway to individuals outside the ORR is the groundwater medium. Leachate from the shallow trenches entering unconfined aquifers below the site would ultimately discharge to the Clinch River. The speed of groundwater movement from the site to the Clinch River (1.5 to 3 km distant) is estimated to be a maximum of 200 to 350 m/d. The aquifer would be diluted by at least a factor of 2.4×10^3 upon complete mixing with the Clinch River flow. The nearest public drinking water supply is located in Kingston, Tennessee, on the Tennessee River, above the confluence with the Clinch River. The Clinch River is, however, a logical future public water supply.

The shallow, unconfined aquifers in the immediate vicinity of the site are not of regional significance. Although water may be obtained from wells in areas that could be contaminated by leachate from the disposal units, the yields of the wells are judged to be relatively low. Hence, except under isolated circumstances, it seems unlikely that shallow aquifers in the vicinity of the disposal units would be tapped for drinking water while more productive water sources are found nearby.

An aboveground disposal option using the tumulus concept is also considered in the pathways analysis since the characteristics of the West Chestnut Ridge Site are amenable to the tumulus concept. This disposal option is considered a major technology alternative to shallow land burial. For the tumulus disposal method, which incorporates a concrete floor, the surface water medium is considered to be the predominant radionuclide migration pathway to persons outside the ORR. Leachate from the tumuli would flow predominantly overland to on-site creeks and ultimately be transported to the Clinch River. The dilution factor for this pathway is estimated to be 5.9×10^4. Ish Creek is the only stream on the site with flow sufficient for establishing a water supply.

In addition to examining these exposure pathways for persons outside the ORR, the potential radiation doses to an inadvertent intruder following site closure and institutional control are also examined. It is assumed that an individual resides on the site, receives direct exposure from the contaminated soil, inhales suspended particles of contaminated dust, ingests vegetables grown on the plot, and consumes contaminated water from either an on-site well (for trench disposal) or a nearby surface stream (for tumulus disposal). This scenario for an inadvertent intruder was used by the Nuclear Regulatory Commission (NRC) to develop a quantitative basis for 10 CFR 61. The NRC assumed that the institutional control period would last for at least 100 years, after which an individual could unknowingly occupy the site and be exposed to waste still containing significant quantities of radioactivity. Although their occurrence on the ORR appears remote, these events are examined for an institutional control period of 100 years and a performance period of 500 years after site closure.

To model groundwater transport of leachate from the shallow trenches, the FEMWATER and FEMWASTE computer codes are used. A two-dimensional, vertical cross section is used to simulate the migration of radionuclides from a trench bottom, through the soil layer (the main buffer layer to radionuclide migration) to the underlying weathered bedrock aquifer. The weathered bedrock zone has a high hydraulic conductivity and affords rapid drainage of infiltrating water to the Clinch River. All of the parameters needed for the model are or based on the characterization study of the site's geologic and hydrologic systems. In all cases, assumptions are

made and parameters are selected to produce conservative (sometimes, worst-case) results. Specifically, the following are noteworthy:

- o No credit is taken for dilution of leachate flow with aquifer flow in the weathered bedrock zone. Site characterization data do not provide sufficient information for a reasonable estimate of the dilution factor.
- o No credit is taken for decay of relatively short-lived radionuclides (e.g., ^{3}H, ^{90}Sr, and ^{137}Cs) during operation of the CWDF, which has an anticipated life span of 40 years.
- o No credit is taken for the form of the wastes to be disposed of at the site.
- o No credit is taken for leachate migration during the 100-year period of institutional control.
- o No credit is taken for the engineered features proposed for the facility or the effectiveness of monitoring and mitigation programs in reducing the migration of contamination.
- o All disposal units are assumed to fail simultaneously after the 100-year institutional control period, resulting in complete saturation of the waste and instantaneous generation of leachate containing radioactivity. The concentrations of radionuclides in the leachate are allowed to range up to the solubility limit for the elements in cases where the limits are known.

Each of these assumptions has the effect of producing a higher concentration of radioactivity in the groundwater than one would expect to find.

Analysis of radionuclide migration in the surface water system for the tumulus disposal option is also based on conservative assumptions. In addition to the assumptions noted above, all of the leachate is assumed to be released directly to the surface water system. An exponential release rate is assumed.

For the waste inventories anticipated from the three Oak Ridge facilities, both trench and tumulus disposal options are found to provide effective containment for persons outside the ORR. The calculated dose commitments are well below limits (25 millirem/year to the whole body or bone) established by the NRC for commercial, low-level radioactive waste

disposal sites. Specifically, the maximum whole-body and bone doses to persons who might use the Clinch River as a source of drinking water are estimated to be 7.9×10^{-2} and 1.05 millirem/year, respectively, for trench disposal and 1.4 and 8.5 millirem/year for tumulus disposal. Trench disposal will provide more effective containment than tumulus disposal because of the retardation effect of the soil on some radionuclides as they migrate through the groundwater system. In view of the conservatisms noted above for the water pathways analysis, there is reasonable assurance that persons outside the ORR would not be exposed to hazardous levels of radioactivity.

The inadvertent intruder, however, could receive radiation doses that exceed (especially for the bone) the 500-millirem/year limit set by the U.S. Department of Energy; this limit was also used by the NRC to establish maximum radionuclide concentrations for disposal of low-level radioactive waste. The water pathway accounts for the bulk of the maximum bone doses for each disposal option: 70% (2,500 millirem/year) for trench disposal and 95% (18,000 millirem/year) for tumulus disposal. For the maximum whole-body dose commitment, the water pathway accounts for 34% and 88%, respectively, for the trench and tumulus disposal options. It is noteworthy that the maximum whole-body and bone doses for trench disposal are less than the 5,000-millirem/year limit for workers at a nuclear facility. Because (1) the occurrence of the intrusion event 100 years after site closure is highly unlikely for the ORR, (2) the conservative assumptions noted above grossly overestimate the dose commitments, and (3) the health risk associated with estimated doses to the intruder are relatively small, both disposal options are judged to provide adequate protection for an inadvertent intruder.

It is concluded that the West Chestnut Ridge Site will provide a suitable location for the CWDF and can be developed to meet appropriate regulations. Independent of waste burial concentrations, the predicted zone of groundwater contamination is defined by Ish Creek to the east, Tennessee Highway 95 and Bear Creek Road to the north, Grassy Creek to the northwest, the Clinch River and the approximate western limit of the subterranean portion of New Zion Creek to the west, and the Clinch River to the south.

ABSTRACT

A dose-to-man pathways analysis is performed for disposal of low-level radioactive waste at the Central Waste Disposal Facility on the West Chestnut Ridge Site. Both shallow land burial (trench) and aboveground (tumulus) disposal methods are considered. The waste volumes, characteristics, and radionuclide concentrations are those of waste streams anticipated from the Oak Ridge National Laboratory, the Y-12 Plant, and the Oak Ridge Gaseous Diffusion Plant. The site capacity for the waste streams is determined on the basis of the pathways analysis.

The exposure pathways examined include (1) migration and transport of leachate from the waste disposal units to the Clinch River (via the groundwater medium for trench disposal and Ish Creek for tumulus disposal) and (2) those potentially associated with inadvertent intrusion following a 100-year period of institutional control: an individual resides on the site, inhales suspended particles of contaminated dust, ingests vegetables grown on the plot, consumes contaminated water from either an on-site well or from a nearby surface stream, and receives direct exposure from the contaminated soil.

It is found that either disposal method would provide effective containment and isolation for the anticipated waste inventory. However, the proposed trench disposal method would provide more effective containment than tumuli because of sorption of some radionuclides in the soil. Persons outside the site boundary would receive radiation doses well below regulatory limits if they were to ingest water from the Clinch River. An inadvertent intruder could receive doses that approach regulatory limits; however, the likelihood of such intrusions and subsequent exposures is remote.

1. INTRODUCTION

A Central Waste Disposal Facility (CWDF) has been proposed for solid low-level radioactive waste generated at the three U.S. Department of Energy (DOE) plants located on the Oak Ridge Reservation: the Oak Ridge National Laboratory, the Y-12 Plant, and the Oak Ridge Gaseous Diffusion Plant. An approximately 506-ha (1250-acre) site on West Chestnut Ridge (Fig. 1.1), bounded by Bear Creek Road to the north, Tennessee Highway 95 on the east, and the New Zion Patrol Road to the south and west, is under consideration for the CWDF. The West Chestnut Ridge Site has undergone comprehensive field studies (Ketelle and Huff 1984) to characterize the geologic and hydrologic systems for shallow land burial, the proposed disposal method for the CWDF. This report presents the results of a pathways analysis investigation undertaken to ascertain the suitability and capacity of the site for the shallow land burial disposal method.

In addition to considering the shallow land burial method of solid low-level radioactive waste disposal, this report also considers the radionuclide migration pathways and attendant dose commitments for an aboveground disposal option using the tumulus concept. Because the characteristics of the West Chestnut Ridge Site are amenable to the tumulus concept, this disposal option was considered a major technology alternative to shallow land burial (DOE 1984).

Methods for obtaining estimates of individual radiation doses likely to be incurred from materials at a low-level waste disposal site have been identified by the Nuclear Regulatory Commission (NRC) in support of 10 CFR 61. Generally, two time periods are of concern: (1) the institutional control period following site closure and (2) the performance period which includes institutional control. For the institutional control period, which is at least 100 years for commercial sites, it is assumed that inadvertent intrusion can be prevented and that any human exposure that may occur will result from off-site migration of the radioactive materials. For commercial sites, the performance period is defined as lasting for at least 500 years or as long as the waste remains hazardous. The NRC has not prescribed a maximum radiation dose to the inadvertent

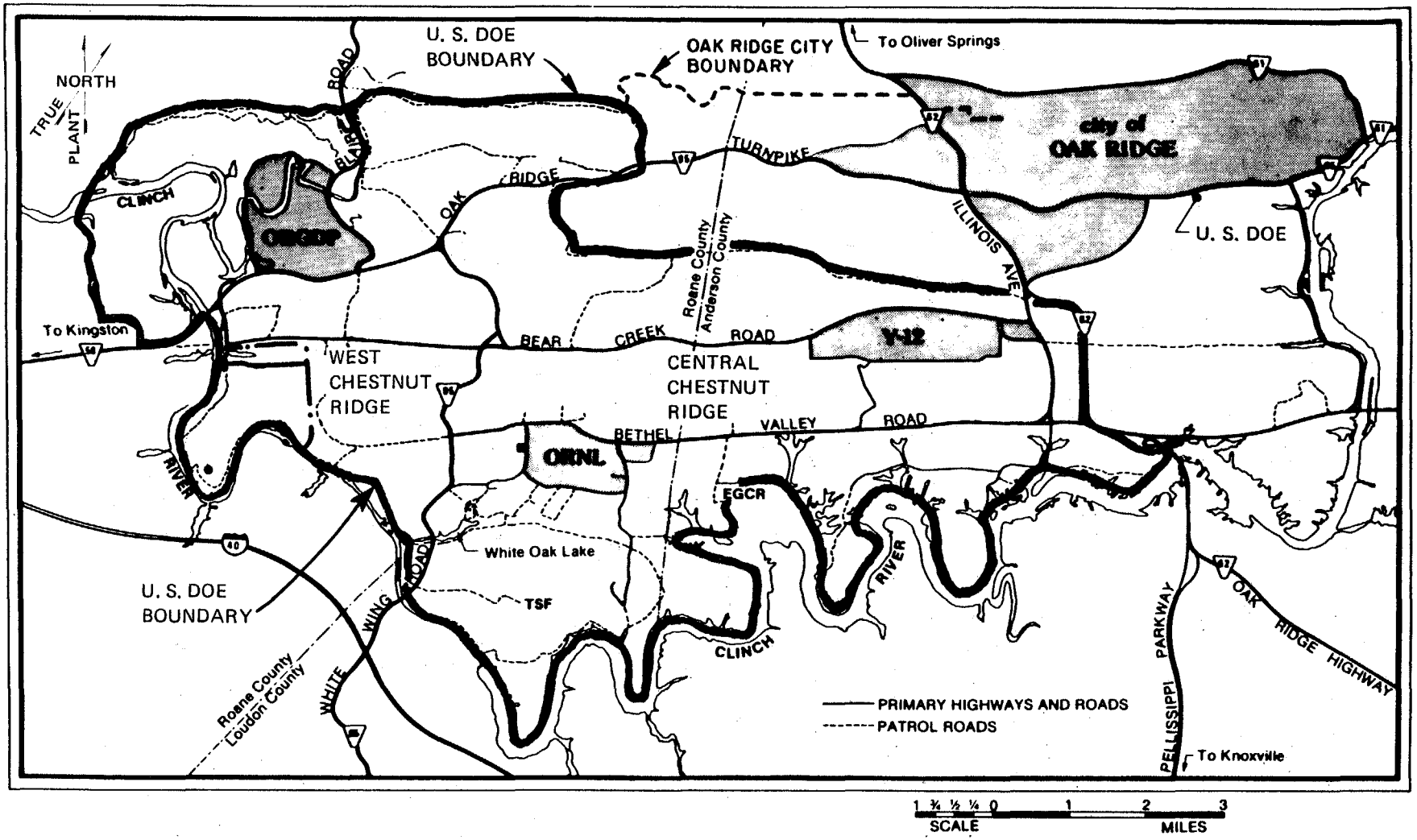


Fig. 1.1. Map of the West Chestnut Ridge Site.

intruder for commercial low-level radioactive waste disposal sites, but a value of 500 millirem/year (whole body) was used in setting maximum radionuclide concentrations for 10 CFR 61. The maximum whole-body dose to individuals outside the site boundary is 25 millirem/year. It is noteworthy that the 500-millirem/year limit is also included in DOE Order 5480.1A and the 25-millirem/year limit has been proposed by the Environmental Protection Agency (40 CFR 191) for disposal of transuranic and high-level wastes. These values are used here as a basis for ascertaining the suitability and capacity of the West Chestnut Ridge Site for disposal of solid low-level radioactive waste anticipated from the three plants.

In this pathways analysis study, the radiation doses to an individual outside the site boundary and to an inadvertent intruder are determined for the radioactive waste inventories expected to be disposed of at the site. These radiation doses are then compared to the maximum permissible doses to obtain the capacity of the site. The zone of groundwater contamination is also determined to establish a buffer zone beyond which human use of the region would be unrestricted. This aspect of the study is essential in determining if the site (disposal units and buffer zone) is of sufficient size.

The pathways considered for the time periods of concern are based on the site and waste characteristics included in Sects. 2 and 3, respectively. The transport of leachate from the disposal units to the Clinch River, the nearest potential source of public drinking water, is analyzed in Sect. 4 for trench disposal (groundwater pathway) and in Sect. 5 for tumulus disposal (surface water pathway). The potential radiation doses for the various pathways are estimated in Sect. 6 and the results are discussed in Sect. 7. Major conclusions for the study are given in Sect. 8.

2. DESCRIPTION OF SITE, FACILITY, AND RADIONUCLIDE MIGRATION PATHWAYS

Extensive field investigations have been performed to characterize the West Chestnut Ridge Site and contiguous area. Results from these investigations have been reported by Woodward and Clyde (1984) and summarized by Ketelle and Huff (1984). In this section, the salient aspects of the site characteristics are discussed. Conceptual designs of the CWDF for the shallow land burial and tumulus disposal options are described. Based on characteristics of the waste, site, and facility designs, the chief radionuclide migration pathways are identified. These data form the basis for the conceptual model of site performance.

2.1 SITE

2.1.1 Topography

The West Chestnut Ridge Site is located near the southwest boundary of the DOE Oak Ridge Reservation (Fig. 1.1). A topographic map of the site is shown in Fig. 2.1. Approximately 60 ha (150 acres) are under consideration for use as a disposal area. The site consists of three discontinuous ridge lines separated by valleys containing the local surface drainage system of first- and second-order streams. Surface runoff drains primarily into Ish Creek and an unnamed ephemeral stream. Smaller portions of the site runoff enter Raccoon Creek to the south and Grassy Creek and Bear Creek to the north. Internal drainage via karst features occurs in several zones on the site.

The site is located in an upland ridge area underlain by southeasterly dipping carbonate bedrock. Active geomorphic processes that are occurring on the site include sheet erosion, localized gully erosion, soil creep on steeper slopes, and subsidence related to dissolution of the carbonate bedrock. A site map showing geomorphic features of the site is presented in Fig. 2.2. Locations of karst features that were observed in field mapping are identified.

Fig. 2.1. Topography of the West Chestnut Ridge Site.

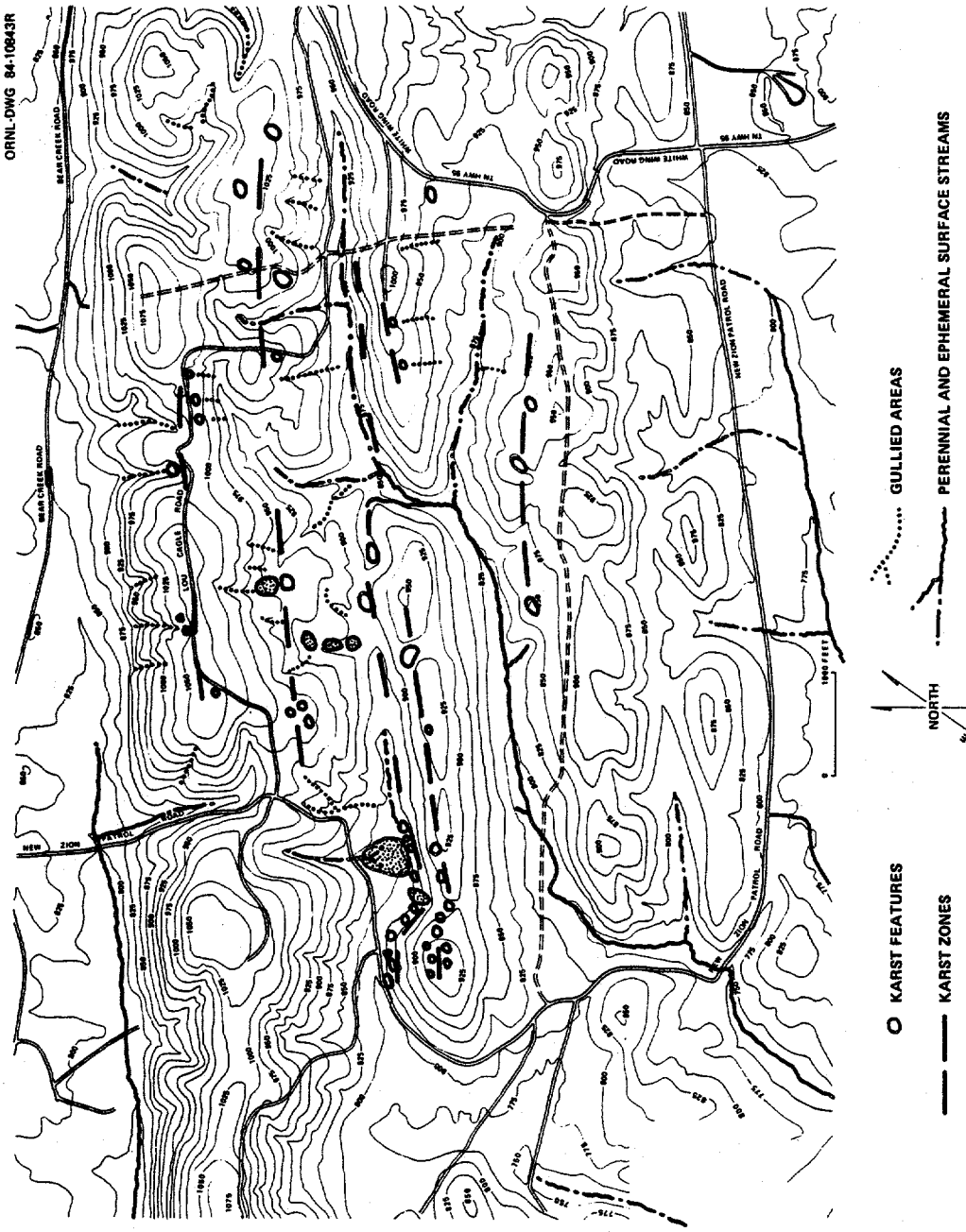


Fig. 2.2. Geomorphic features on the West Chestnut Ridge Site.

2.1.2 Geology

The bedrock units that underlie the site are predominantly carbonate rocks in the Conasauga and Knox groups. A geologic profile of the site is shown in Fig. 2.3, and an areal map is shown in Fig. 2.4. The mean bedding strike is N57°E with a mean bedrock dip of 31° to the southeast. The analysis of the structural geology shows the bedrock fracture orientations to be of four generalized groups: (1) parallel to strike, (2) perpendicular to strike, (3) north-south, and (4) east-west.

Residual soils of variable thickness overlie the bedrock. A rubbly weathering zone occurs between the residuum and sound bedrock. This zone ranges from 0 to >30 m thick and consists of cavitose rock with mud- and gravel-filled cavities. The approximate top of the zone of cavitose bedrock is shown in Fig. 2.5. The approximate top of continuous rock is shown in Fig. 2.6.

2.1.3 Soils

The site soils are fine grained and are classified predominantly as highly plastic clays (CH) with traces of fine to coarse sand and chert gravel. These soils have high moisture retention properties and are typically greater than 90% saturated below depths of 3 m. Surficial soils are predominantly silts and sands with variable chert gravel contents. Fe/Mn nodules are typically present in the soils. Surficial clays are predominantly kaolinite and hydroxy interlayered vermiculite with variable amounts of mica, vermiculite, and quartz. The soil pH has a range of 4.1-6.0. Residual soils at depth are predominantly clays, composed largely of kaolinite with smaller fractions of mica and vermiculite. Gravel contents are variable and dominated by chert. The pH of the residuum ranges between 5.0 and 6.7.

Geochemical analyses of site soils indicate that the residual soils have been leached extensively and are acid pH with low carbonate content. These soils overlie a zone influenced by the presence of weathered carbonate rock having neutral to alkaline pH and higher calcium, magnesium, and soluble carbonate contents. These two zones undoubtedly interfinger extensively in the weathered bedrock zone. Sorption and desorption

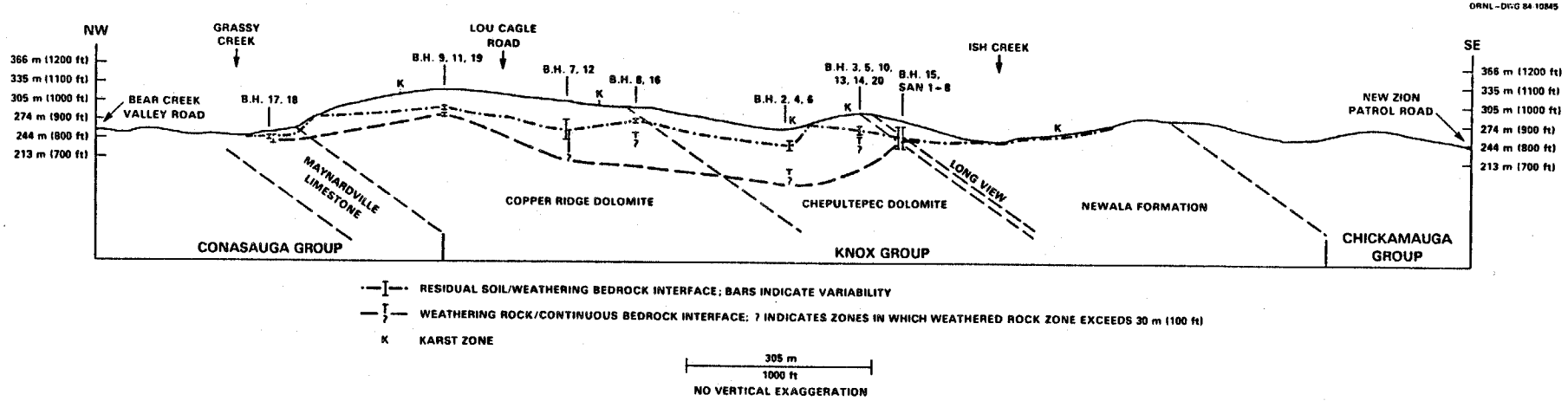


Fig. 2.3. Generalized geologic profile through the West Chestnut Ridge Site.

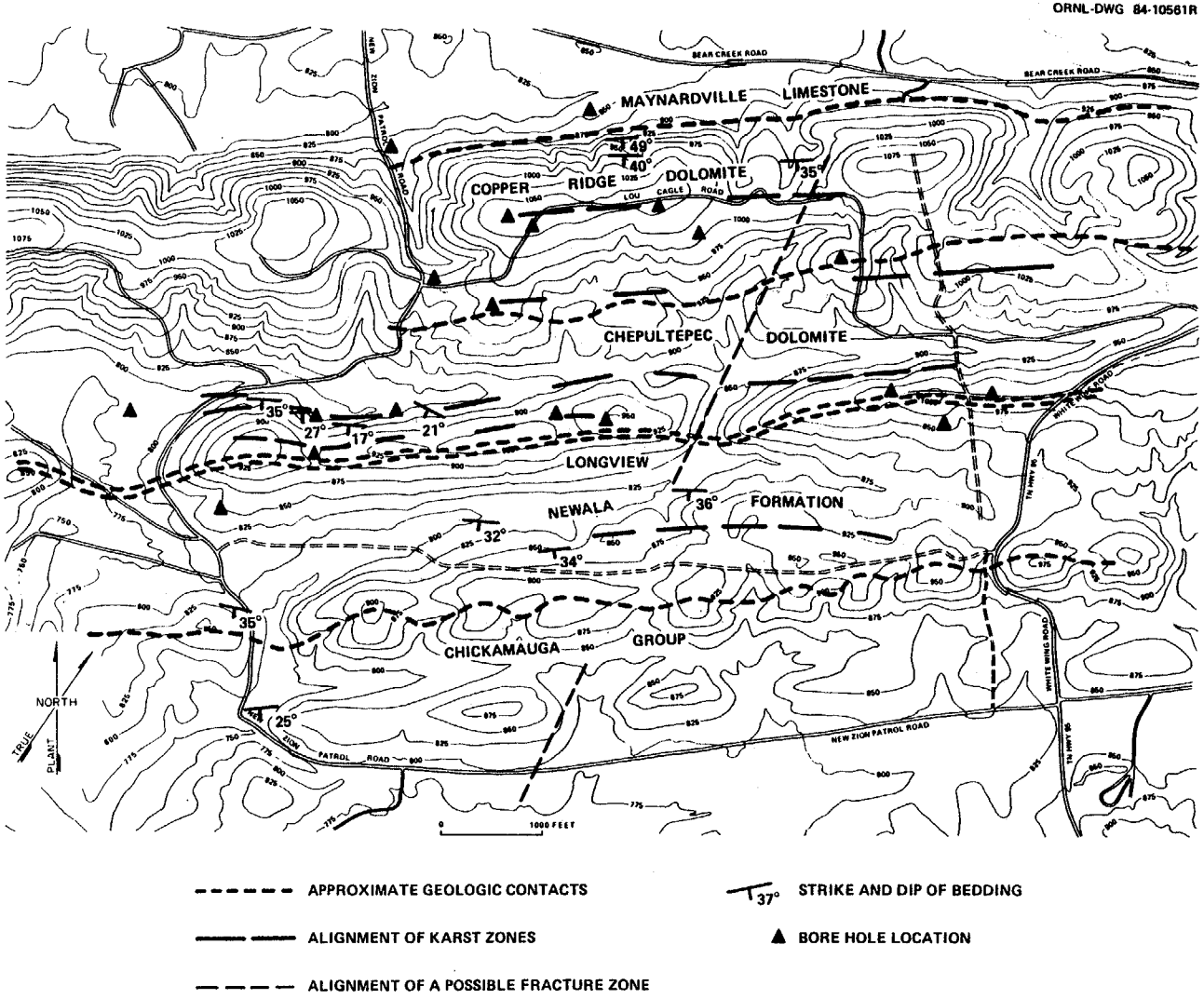


Fig. 2.4. Areal geologic map of the West Chestnut Ridge Site.

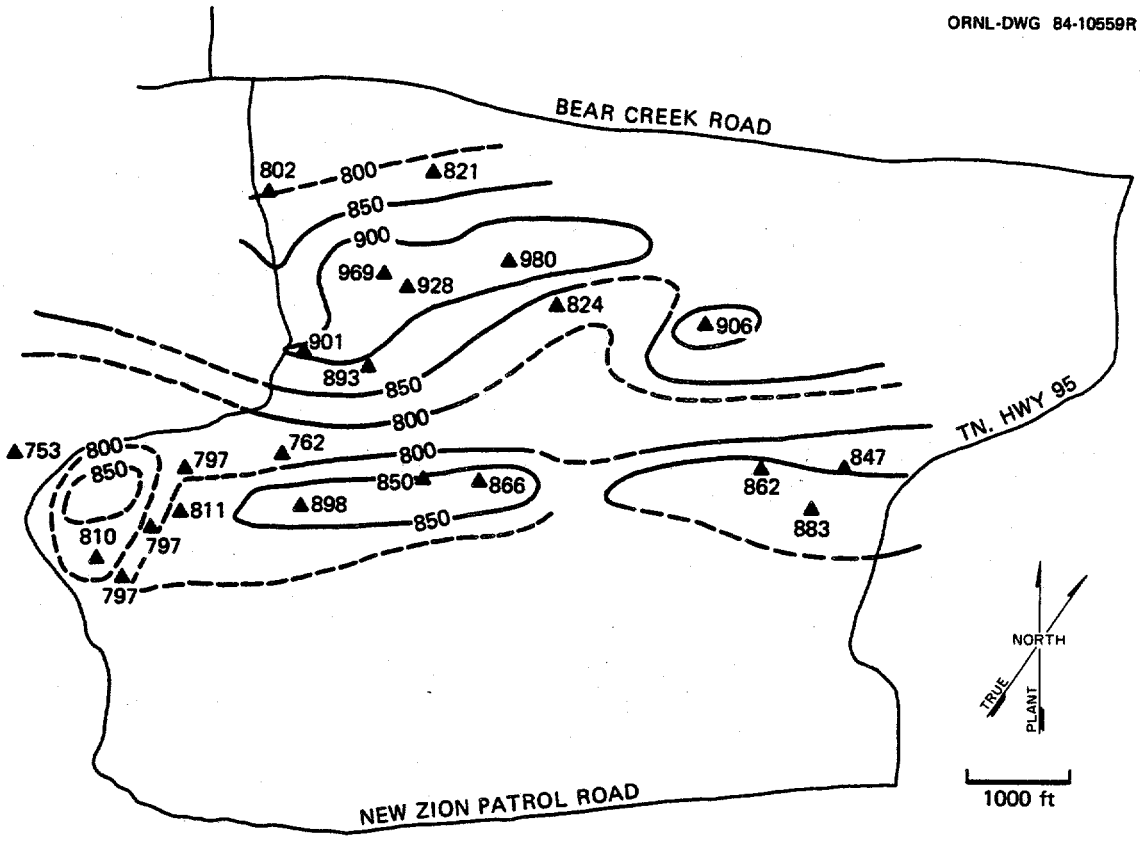


Fig. 2.5. Approximate configuration of the top of the weathered bedrock zone.

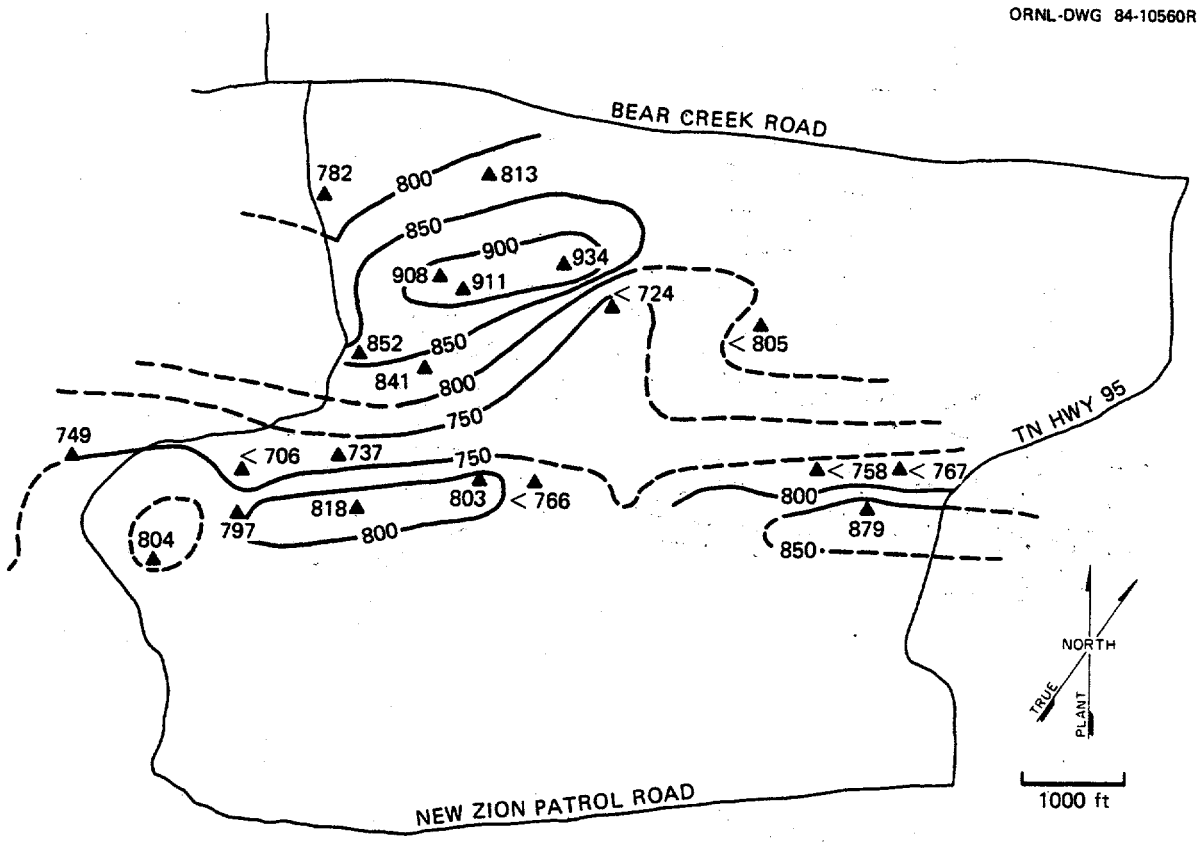


Fig. 2.6. Approximate configuration of the top of continuous bedrock.

characteristics of Knox residuum have been investigated by Seeley and Kelmers (1984), and the results are summarized in Table 2.1. Calculations of the solubility of radionuclides in site soils and groundwater are discussed in Appendix A.

The existence of very gentle slopes on the site, which are typical of karst features, suggests either that they have formed by gradual downwarping or that they are very old features. Considering the apparent age and topographic character of the Knox residuum, the karst movement at the site is interpreted to be largely the result of plastic deformation.

2.1.4 Geohydrology

Unconfined aquifers occur in the soil, weathered bedrock, and bedrock zones at the site. The flow is from the higher to lower topographic elevation with gradients indicating flow towards the nearest surface water features. In the Copper Ridge Dolomite outcrop belt, the apparent groundwater divide coincides with the topographic divide; however, in the Longview/Newala Ridge, the apparent divide occurs approximately 100 m southeast of the ridge crest. Transmission of water through soil and bedrock occurs rapidly. The vertical infiltration capacity of the soil typically is exceeded during the winter and spring seasons, resulting in lateral quickflow in the upper soil horizons and saturated wetting fronts infiltrating through the soils. The water movement in the weathered bedrock and bedrock is strongly controlled by solution cavities associated with bedding orientations, penetrative joints, and fractures. The flow paths are interpreted as trellis drainage patterns with long runs parallel to strike and cross-strike channels leading to the ground surface or other strike-controlled zones.

The saturated permeability of the soils at the site is summarized in Table 2.2. The data show a generalized tendency to decrease with depth and to have less data scatter with depth. The permeability of the weathered bedrock zone was determined to be 1.7×10^{-4} cm/s with a transmissivity of 1.7×10^{-1} cm^2/s. A representative value of the permeability of the unweathered bedrock is 1.0×10^{-4} cm/s in the transmissive zones and zero in the nontransmissive zones. The permeability in the unweathered rock is attributed to flow in fractures and/or open bedding planes.

Table 2.1. Summary of radionuclide sorption and desorption data

Radionuclide (valence)	pH	R_s values[a]	
		Range	L/kg
U(+6) (UO^+)	5.6 ± 1.0	Average	3.2×10^3
		High	2.5×10^4
		Low	2.5×10^2
Sr(+2)	6.0 ± 0.6	Average	6.9×10^2
		High	1.6×10^3
		Low	2.0×10^2
Cs(+1)	5.4 ± 0.7	Average	3.3×10^3
		High	1.1×10^4
		Low	1.1×10^2
Co(+2)	6.0 ± 1.0	Average	1.6×10^3
		High	7.9×10^3
		Low	7.1×10^1
Eu(+3)	5.0 ± 0.7	(one only)	
		High	6.1×10^4
		Low	6.4×10^1
Th(+4)	4.0 ± 0.7	(one only)	
		High	1.1×10^4
		Low	5.4
Tc(-1)	5.1 ± 0.2	(one only)	
		High	1.6
		Low	1.0
I(-1)	5.8 ± 0.6	Average	1.8×10^{-1}
		High	1.8
		Low	1.4×10^{-2}

[a] R_s values are derived from contacts with low initial concentrations of the radionuclide (5 mg/L).

Source: Seeley, F. G., and A. D. Kelmers. 1984. Geochemical Information for the West Chestnut Ridge Central Waste Disposal Facility for Low-Level Radioactive Waste. ORNL-6061, Oak Ridge National Laboratory.

Table 2.2. Summary of field and laboratory saturated permeability test results

Typical depth and type of test	Mean permeability (cm/s)	Mean ± 1 standard deviation (cm/s)
6 m (20 ft)		
- Field tests	6.1×10^{-6}	7.9×10^{-5} to 5.0×10^{-7}
- Lab tests	3.2×10^{-6}	7.9×10^{-5} to 1.3×10^{-7}
12 m (40 ft)		
- Field tests	2.0×10^{-6}	2.0×10^{-5} to 2.0×10^{-7}
- Lab tests	1.0×10^{-7}	2.5×10^{-7} to 4×10^{-8}
21-30 m (70-100 ft)		
- Field tests	No data	No data
- Lab tests	6.3×10^{-8}	5.0×10^{-8} to 7.9×10^{-8}

Source: Ketelle, R. H., and D. D. Huff. 1984. Site Characterization of the West Chestnut Ridge Site. ORNL/TM-9229, Oak Ridge National Laboratory.

The unsaturated permeability and moisture characteristics of the site soils were determined by Daniels and Broderick in Appendix S of Woodward and Clyde (1984). The moisture-suction test results are summarized in Fig. 2.7, and the relative permeability-suction results are summarized in Table 2.3. The relative permeability at any suction is multiplied by the saturated permeability to obtain the unsaturated permeability.

Water levels in the soil and bedrock showed seasonal fluctuations of 3-15 m; however, not all wells showed seasonal fluctuations. Wells screened in soils tend to respond to precipitation events more rapidly than bedrock wells. Both bedrock and soil wells in topographically low areas respond rapidly to precipitation events. The upland portions of the site indicate the presence of two saturated zones, one in the soil and one in the weathered bedrock and bedrock. The fluctuations of the water levels in the two zones vary widely. The hydrographs for bedrock wells in the same stratigraphic intervals show similar behavior. The hydrographs for the wells on the site are reported by Ketelle and Huff (1984). In the lower elevations of the site, maximum water table elevations are within 1 m of the ground surface. Figures 2.8 and 2.9 show the relationship between the topographic location and the maximum observed water elevations that occurred in May 1984.

A dye tracer test performed on the site in the bedrock aquifer showed velocities of the order of 240-380 m/d from the site to the Clinch River 1.5-3 km away. These results are considered to represent an upper bound of groundwater movement for the site.

2.1.5 Surface Water

The site drainage discharges to two perennial streams, Ish Creek to the south and Grassy Creek to the north, and to an ephemeral stream in the middle of the site, which discharge to the Clinch River. The discharges of the creeks were monitored intermittently from July 1982 to July 1983 and continuously from November 1983 to April 1984 at the sites identified in Fig. 2.10. The summary of the data is presented in Tables 2.4 and 2.5. The average flow of the Clinch River at Melton Hill Dam, which is located approximately 5 km from the site, is 150 m^3/s. The flows in the streams

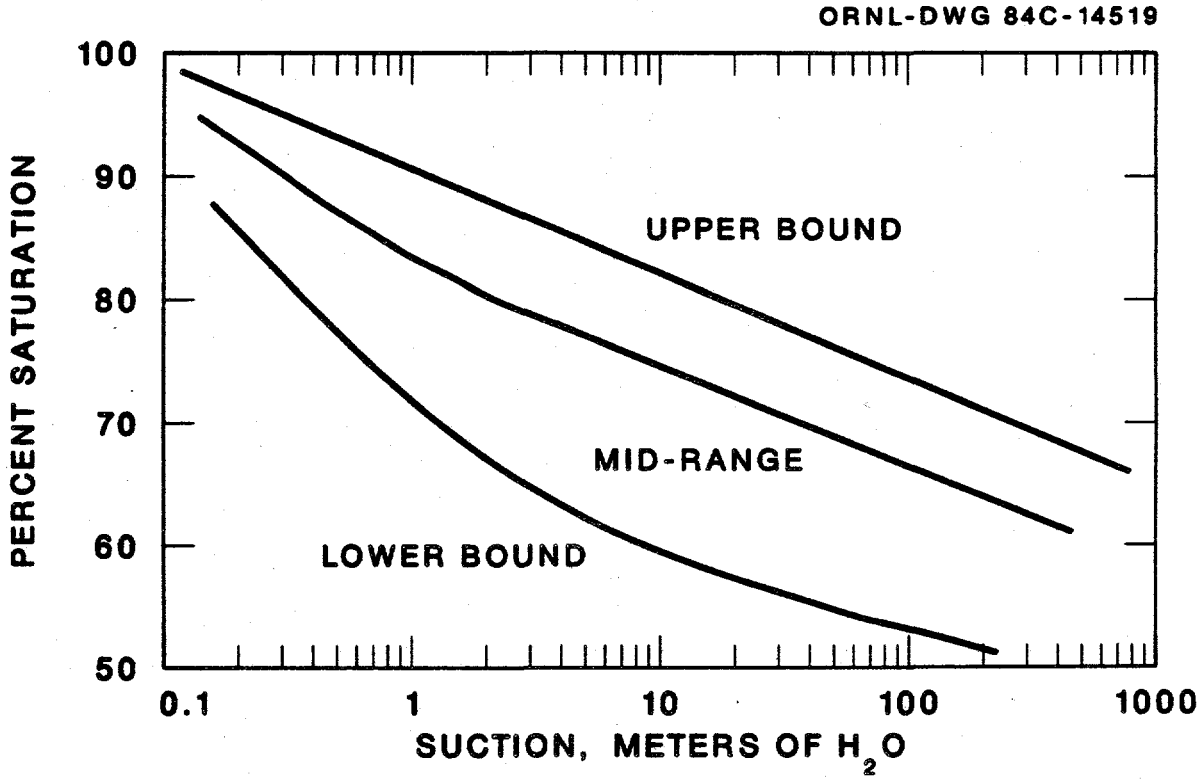


Fig. 2.7. Recommended moisture-suction curves.

Table 2.3. Recommended range of values for curves of relative permeability versus soil suction

Soil suction (meters of water)	Relative permeability		
	Upper bound	Mid-range	Lower bound
0	1	1	1
0.1	6×10^{-1}	2×10^{-1}	4×10^{-1}
0.2	5×10^{-1}	1×10^{-1}	8×10^{-2}
0.5	3×10^{-1}	4×10^{-2}	1×10^{-2}
1	2×10^{-1}	2×10^{-2}	3×10^{-3}
2	1×10^{-1}	1×10^{-2}	9×10^{-4}
5	4×10^{-2}	4×10^{-3}	1×10^{-4}
10	2×10^{-2}	1×10^{-3}	4×10^{-5}
20	1×10^{-2}	5×10^{-4}	1×10^{-5}
50	6×10^{-3}	1×10^{-4}	2×10^{-6}
100	3×10^{-3}	8×10^{-5}	5×10^{-7}
200	8×10^{-4}	2×10^{-5}	1×10^{-7}
500	1×10^{-4}	5×10^{-6}	2×10^{-8}

Source: Ketelle, R. H., and D. D. Huff. 1984. Site Characterization of the West Chestnut Ridge Site. ORNL/TM-9229, Oak Ridge National Laboratory.

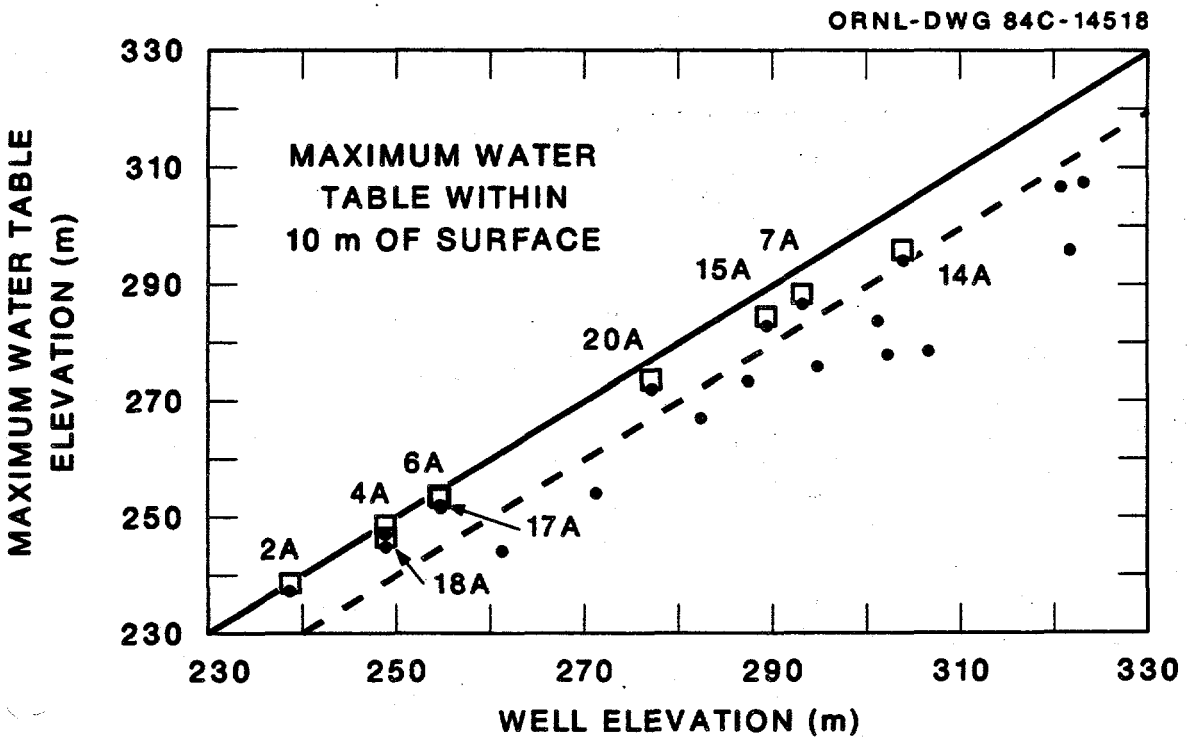


Fig. 2.8. Well elevation versus maximum water table elevations, soil wells.

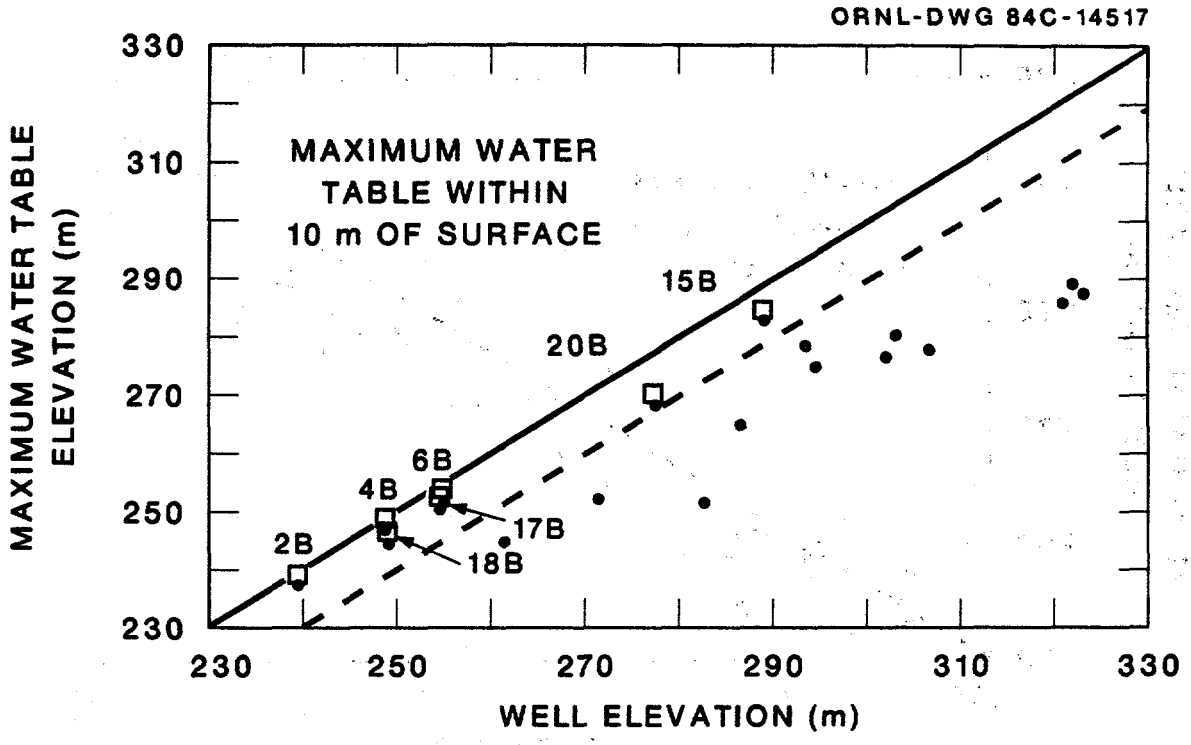


Fig. 2.9. Well elevation versus maximum water table elevations, bedrock wells.

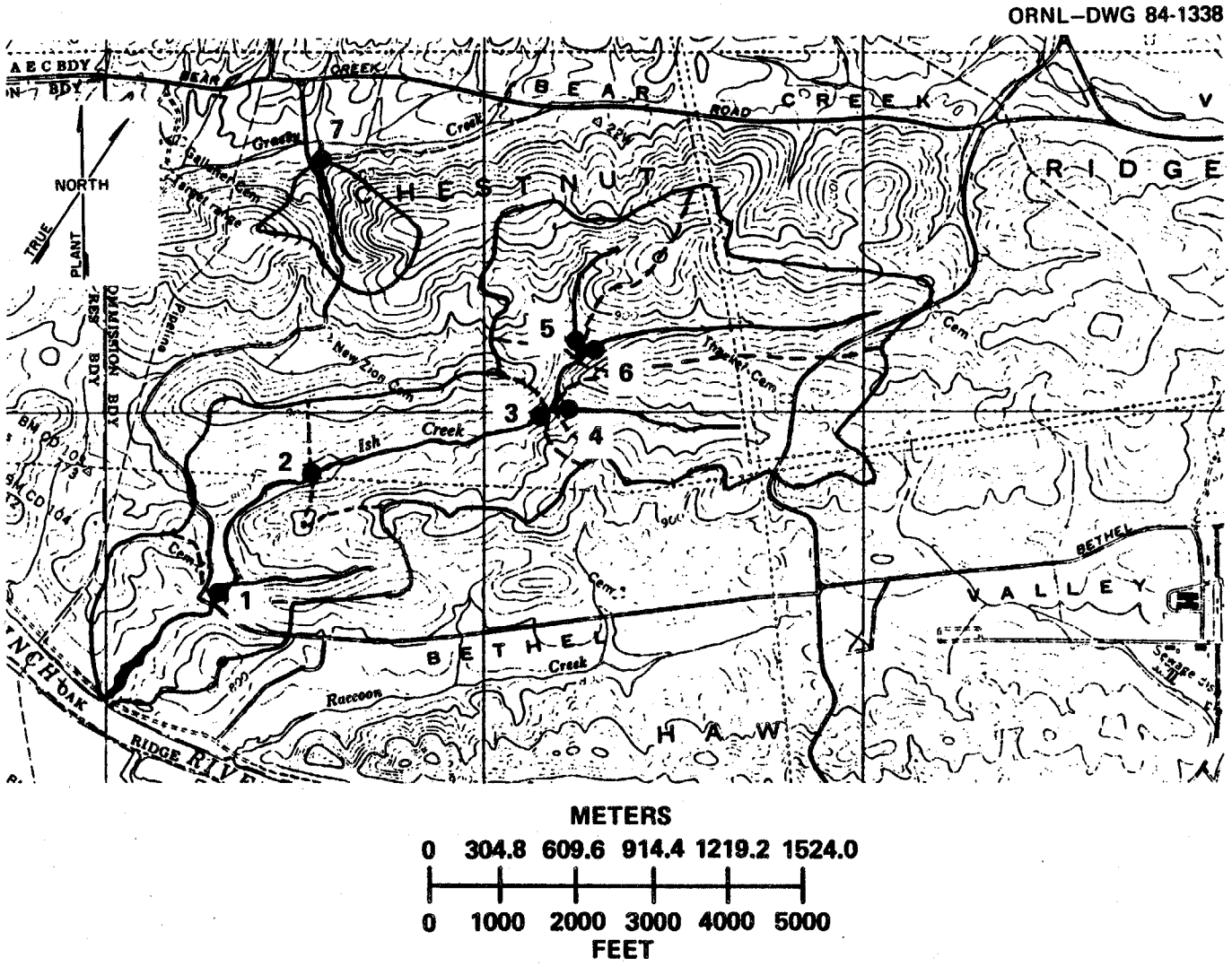


Fig. 2.10. Drainage area for Ish Creek and location map for seven temporary streamflow monitoring stations.

Table 2.4. Monthly flow data for monitoring stations on the West Chestnut Ridge Site (Units are L/s)

	Nov. 1983	Dec. 1983	Jan. 1984	Feb. 1984	Mar. 1984	Apr. 1984
CWDF 1						
Monthly mean	23.7	93.2	45.4	50.1	73.2	49.6
Instantaneous max	1750	1010	132	346	682	281
Instantaneous min	1.2	14	9.6	9.6	14	14
CWDF 3						
Monthly mean	6.82	41.7	21.8	25.6	39.4	32.5
Instantaneous max	185	283	71	150	318	130[a]
Instantaneous min	0.6	5.7	4.4	3.2	6.4	8.1
CWDF 4						
Monthly mean	2.23	14.9	8.55	10.1	12.0	8.30
Instantaneous max	>48	>48	22	>48	>48	>48
Instantaneous min	0.03	2.8	2.3	1.5	2.7	1.8
CWDF 7						
Monthly mean	0.68	3.82	2.50	2.62	4.05	3.33
Instantaneous max	10	21	7.2	14	31	13
Instantaneous min	0.1	0.8	0.7	0.7	1.0	1.8
CWDF 8[b]						
Monthly mean	0.87	7.11	4.16	4.97	8.50	6.79

[a]Estimated.

[b]Fragmentary stage record at CWDF 8 is adequate to allow reasonable estimates of mean daily discharge, but except for NO FLOW prior to Nov. 27, instantaneous maximum and minimum values cannot be reliably determined. Maximum flow probably exceeded 48 L/s on March 28.

Source: Ketelle, R. H., and D. D. Huff. 1984. Site Characterization of the West Chestnut Ridge Site. ORNL/TM-9229, Oak Ridge National Laboratory.

Table 2.5. Summary of intermittent flow measurements data for the period July 15, 1982, to July 11, 1983.

Station	Contributing area (km^2)	Annual means flow rates		Maximum flow rate		Minimum flow rate	
		L/s	cm	L/s	cm/d	L/s	cm/d
1	2.44	38.9	50.3	139.0	0.49	1.33	0.005
2	1.94	21.8	35.4	78.7	0.35	0.82	0.004
3	1.45	14.0	30.4	49.7	0.30	0.33	0.002
4	0.54	3.2	18.7	10.9	0.17	0.00	0.000
5	0.25	1.9	24.0	6.8	0.24	0.32	0.011
6	0.52	3.9	23.6	14.9	0.25	0.00	0.000
7	0.14	1.6	36.0	4.4	0.27	0.39	0.024

Source: Ketelle, R. H., and D. D. Huff. 1984. Site Characterization of the West Chestnut Ridge Site. ORNL/TM-9229, Oak Ridge National Laboratory.

on the site are insufficient to support a public drinking water supply but could be sufficient for an individual at Stations 1, 2, or 3. The Clinch River is currently not used for a public drinking water supply downstream of the site; however, Clinch River water quality is good enough to be developed as a water supply in the future.

2.2 FACILITY

The layout of the CWDF--including disposal areas, ancillary facilities, and some access roads--is shown in Fig. 2.11. The completed facility would consist of an operations area and three major disposal areas. At the time of closure of the CWDF, the total required capacity of the facility would be about 6×10^5 m^3 (2×10^7 ft^3). Although the proposed design of the disposal facility consists of shallow trenches (Ebasco 1984), the site can be developed to accommodate tumuli. The site layout would generally be the same for both disposal options.

2.2.1 Shallow Land Burial (Trench) Disposal Units

Two basic trench designs will be used at the CWDF. One design (Fig. 2.12) will be initially 46 m (150 ft) wide at the top, 14 m (45 ft) wide at the bottom, and 9.1 m (30 ft) deep, with a waste layer 6.7 m (22 ft) thick. Its length would be typically 107 m (350 ft), with 91 m (300 ft) available for storage--except where site geometry dictates otherwise. The other trench design, to be used for disposal of the wastes containing asbestos, has smaller dimensions. It would be typically 15 m (50 ft) wide at the top, 3.0 m (10 ft) wide at the bottom, and 5.5 m (18 ft) deep, with a waste layer 3.0 m (10 ft) thick. The length would vary but would be typically 21 m (70 ft). These sizes are given to characterize a reference trench; in practice, trenches would be constructed with minor variations in any of these dimensions, and it is expected that such variations would not significantly affect performance. Trench dimensions might vary in response to operating experience and variation in waste quantitites delivered.

The overburden layer will be 2.4 m (8 ft) thick. The side walls of all trenches will be lined with a drain matting. This material is

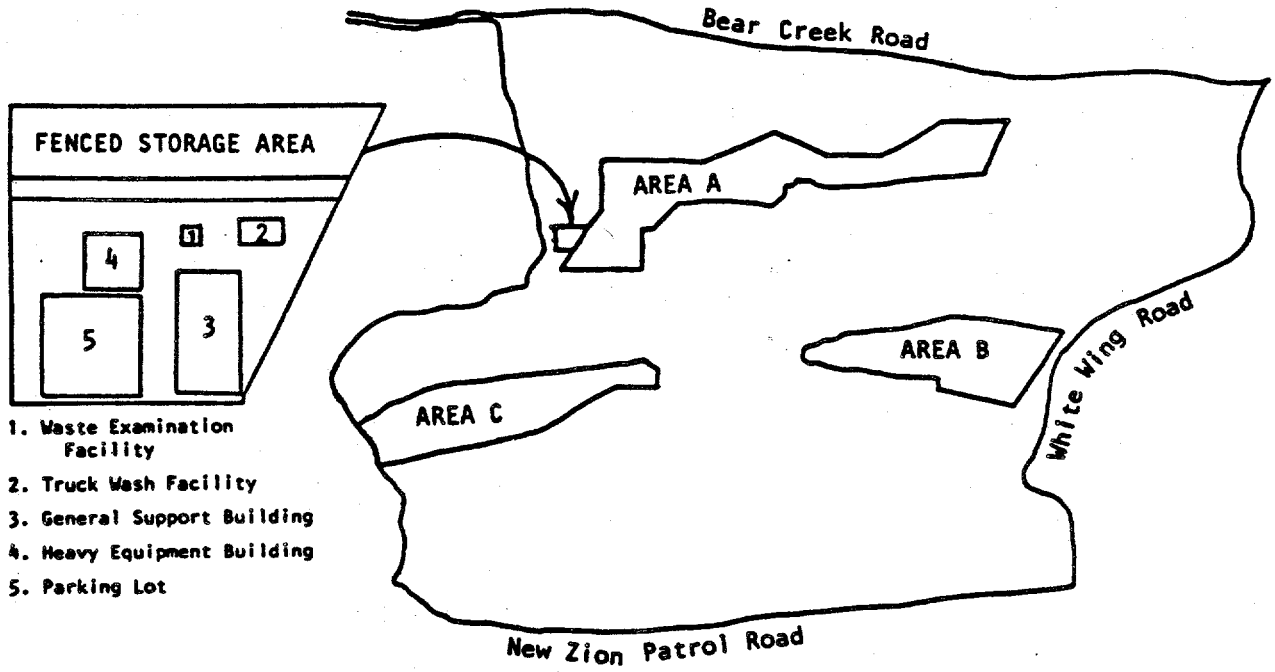
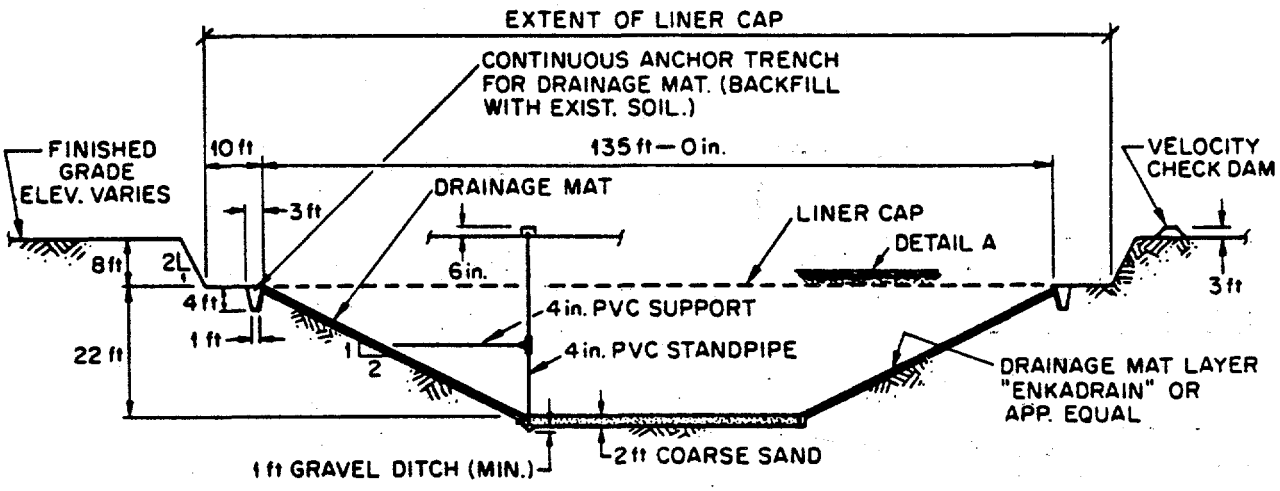
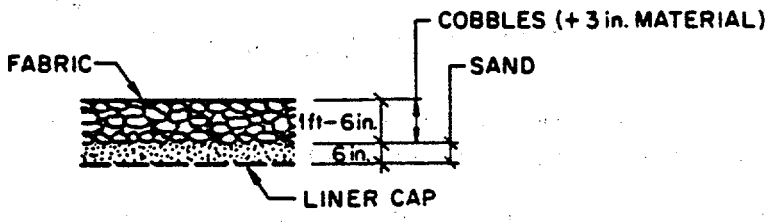


Fig. 2.11. Layout of the Central Waste Disposal Facility. Three major disposal units would be located in Areas A, B, and C.

Source: Ebasco Services, Inc. 1984. Conceptual Design Report for Central Waste Disposal Facility.



SECTION A



DETAIL A

NOT TO SCALE

Fig. 2.12. Schematic diagram of the trench design.

Source: Ebasco Services, Inc. 1984. Conceptual Design Report for Central Waste Disposal Facility.

available in sheet form on rolls with filter fabric on both faces. The purpose of this mat is to establish a capillary break between the soil and the waste.

The trench floor will be designed to collect any water that enters the trench during waste emplacement, permit monitoring after closure, and avoid the bathtub effect. A trench drainage system will be designed to direct any water that enters the trench to a French drain and eventually to a corner sump. The French drain--a gravel-filled, V-shaped tunnel running along the lower longitudinal side of the bottom of a trench--will drain to a sump at the lower end of the trench. Polyvinyl standpipes of 10-cm (4-in.) diameter will connect the surface with the French drain and allow sampling of the drain and monitoring of the movement of isotopes. It would be possible to drain the sump on each trench with a pump.

Each trench, after it has been backfilled, will be covered with an impermeable membrane. This membrane will be covered with a protective layer and with a drain. The trench would be topped with a 1.8-m (6-ft) layer of compacted soil. The topsoil layer will have a vegetative cover to control erosion and to reduce loss of soil moisture.

2.2.2 Tumulus Disposal Units

An alternative to the below-grade (near-surface) trench is an above-grade tumulus structure (DOE 1984). A tumulus is an artificial hillock or mound; thus, each finished disposal unit would be a mound, rising about 9 m (30 ft) above the surrounding land. This concept has been successfully operated for several years at the Centre de la Manche, France.

The design of the above-grade disposal unit is illustrated in Fig. 2.13. The unit would have a concrete floor. and the walls would consist of stacked, cylindrical, concrete blocks. These blocks would have been previously cast from mixtures of cement and either pond sludges (Sect. 3) or uncontaminated aggregate and allowed to cure. The wastes would be piled on the concrete floor and surrounded with gravel. A complete unit would have a layered cap to provide stability and prevent infiltration of water.

The concrete blocks could be isolated from underlying gravel and soil. The blocks would be in contact with only the drained floor and other

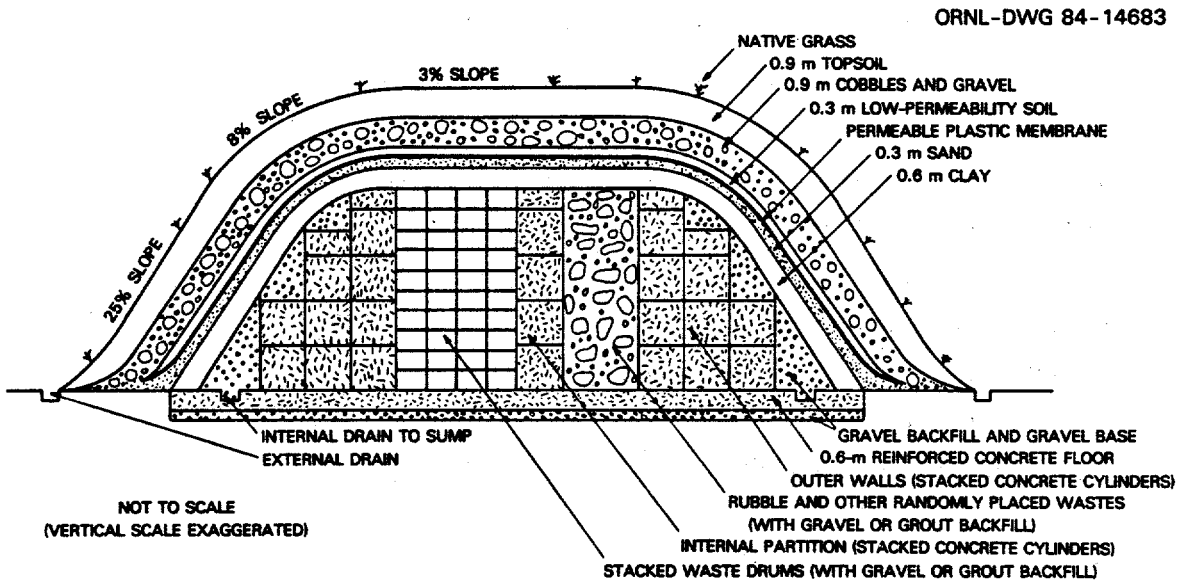


Fig. 2.13. Schematic diagram of the alternative above-grade disposal design.

components of the cap. The concrete used to form the footings and floor, which would be in contact with the underlying gravel and soil, would be prepared from commercial cement and noncontaminated aggregate.

2.3 RADIONUCLIDE MIGRATION PATHWAYS

Pathways of interest for low-level radioactive waste disposal sites are (1) inadvertent intrusion into the waste and its subsequent intake by inhalation and ingestion and (2) groundwater and surface water transport of leachate from the waste and subsequent use of the contaminated water for irrigation and drinking (NRC 1982, Gilbert et al. 1983). As illustrated in Fig. 2.14, wind and water erosion are also processes of transport that can result in environmental exposures. The intruder pathway can occur after the end of institutional control and involve either direct contact with the waste (e.g., someone searching for artifacts) or indirect contact (e.g., agricultural activities). In general, intruder exposure pathways depend on and limit the maximum concentration of the radionuclides in the buried waste and tend to be individual-restrictive and not site-specific. Conversely, groundwater and surface water migration of leachate from the waste (Fig. 2.14) depends on site-specific parameters and tends to limit the total radionuclide quantity disposed of.

2.3.1 Direct Intrusion

Exposure pathways for an inadvertent intruder could result from (1) direct exposure to contaminated soils, (2) inhalation of contaminated dust particles suspended in air by various activities, (3) ingestion of food crops grown in contaminated soil, and (4) ingestion of contaminated surface water or groundwater near the waste disposal area. The first three pathways are potentially realistic in the immediate vicinity of the waste disposal landfill and could occur for either disposal option. However, these pathways are more realistic for the tumulus, because it is less likely to maintain isolation over the long term as compared to shallow land burial. The relatively steep side slopes of the tumulus (Sect. 2.2.2) suggest an increased potential for erosion.

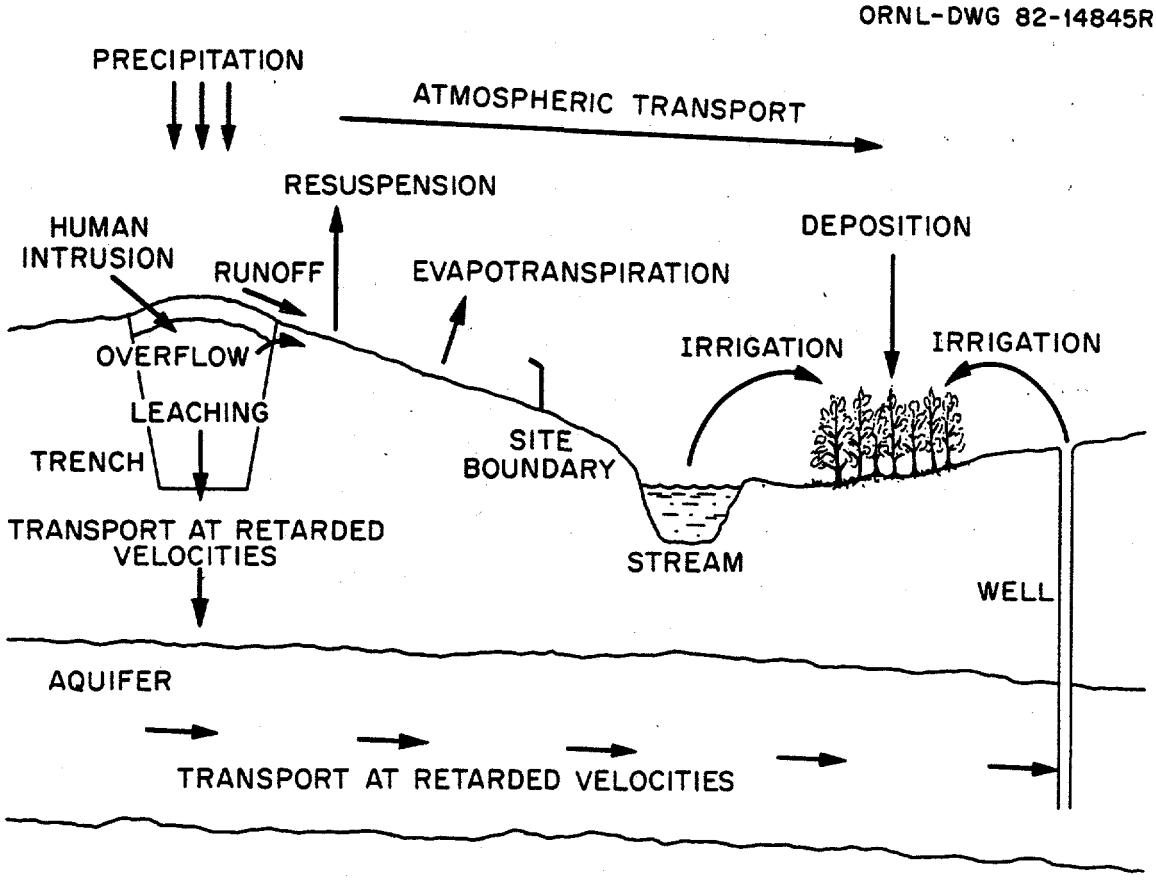


Fig. 2.14. Schematic representations of potential pathways of radiological exposure to humans from waste disposal sites.

An intruder may encounter contaminated water in an on-site well or in a surface stream. For waste disposal in shallow trenches, contaminated groundwater is the anticipated pathway for migration in water; for disposal in tumuli, contaminated surface water is the anticipated pathway for migration in water.

Applicable standards for protection of inadvertent intruders are contained in DOE Order 5480.1A (DOE 1981). Specifically, for uncontrolled areas, the maximum dose to an individual should not exceed 500 millirem/year to the whole body, gonads, or bone marrow and 1500 millirem/year to other organs. Thus, the cumulative radiation dose from all intruder pathways should not exceed these limits.

2.3.2 Groundwater

The groundwater medium affords the predominant radionuclide migration pathway for low-level waste disposal in shallow trenches as designed for the CWDF (Sect. 2.2.3). It provides the most likely exposure pathway for migration of leachate from the disposal units to individuals outside the site boundary and to inadvertent intruders. Leachate can enter the shallow, unconfined aquifers in the immediate vicinity of the West Chestnut Ridge Site, which ultimately discharge to the Clinch River. The nearest public drinking water supply is located in Kingston, Tennessee, on the Tennessee River above the confluence with the Clinch River. The Clinch River is, however, a dependable water supply and is a logical future public water supply. Thus, the concentration of radioactive materials that enter the Clinch river from the CWDF should be sufficiently low to limit the radiation dose to 25 millirem/year to the whole body, 75 millirem/year to the thyroid, and 25 millirem/year to any other organ of an individual.

An inadvertent intruder (Sect. 2.3.1) could obtain drinking water from a well drilled on the West Chestnut Ridge Site. The concentration of radioactive material (leachate) in the water would depend on the location of the well. The site characterization data (Ketelle and Huff 1984) provide evidence that water may be obtained from wells in the site vicinity but that the wells could have a comparatively low yield if they were drilled near the trenches in the weathered bedrock zone. Hence, except

under isolated circumstances, it seems unlikely that shallow aquifers in the vicinity of the disposal units would be tapped for drinking water while more productive water sources are found nearby. Although the groundwater pathway for the inadvertent intruder considers exposure of an individual at a point of maximum exposure, it should be kept in mind that the probability of such an event is low.

2.3.3 Surface Water

The surface water medium is the predominant exposure pathway for tumulus disposal. Leachate from the tumulus would flow primarily overland to the creeks located on site and ultimately be transported to the Clinch River, which is a logical future public water supply. Thus, a potential exists for exposure of individuals outside the site boundary for waste disposal in tumuli. An inadvertent intruder could also use on-site streams for drinking water and be exposed to radioactive materials from the waste disposal units. Ish Creek (at Stations 1, 2, and 3) is the only stream on the site with flow capable of establishing an individual water supply (Sect. 2.1.5).

3. WASTE CHARACTERISTICS

This section discusses the characteristics of waste to be disposed of at the CWDF. Assumptions made for inclusion of the waste streams in the pathways analysis are also discussed. A detailed characterization of the waste streams is given in Appendix B.

3.1 ANTICIPATED WASTE STREAMS

The CWDF will serve as a repository for a variety of radionuclides. A listing of the radionuclides included in the various waste streams is given in Table 3.1. The group composition will be fixed by the research, development, and production activities conducted at the Oak Ridge National Laboratory, the Y-12 Plant, and the Oak Ridge Gaseous Diffusion Plant. Although the relative amounts of radionuclides contained in wastes designated for disposal in the CWDF will change as plant programs change, the spectrum of radionuclides will remain approximately constant because of the characteristics of the DOE programs that are supported by plant activities.

Wastes from the three plants occur in a variety of forms such as laboratory trash, sludges or soils fixed in grout, and demolition and construction materials from the decommissioning of projects and buildings. These waste forms are presented here in three categories corresponding to the categories for emplacement in separate disposal trenches: (1) solid debris (bulk waste), (2) waste contaminanted with asbestos, and (3) grout waste resulting from fixation of sludges and soils. The rate of waste disposal for the first four years is expected to be about 2×10^4 m^3/year (6×10^5 ft^3/year) for grout and 8×10^3 m^3/year (3×10^5 ft^3/year) for other wastes. After the first four years, the disposal rate would be 6×10^3 m^3/year (2×10^5 ft^3/year) for grout and 8×10^3 m^3/year (3×10^5 ft^3/year) for other wastes. The waste volumes presented here are the best estimates available at this time from the waste-contributing plants. Changes in these volumes might result from changes in plant programs or in methods of treating some waste forms.

Table 3.1. Radionuclides contained in waste streams to be disposed of at the Central Waste Disposal Facility

Radionuclide	Emplacement Category		
	Solid debris (bulk waste)	Groute Waste	Asbestos-contaminated waste
^{3}H	x		
^{14}C	x		
^{60}Co	x		
^{63}Ni	x		
^{90}Sr	x		
^{93}Zr	x	x	
^{99}Tc	x	x	
^{121m}Sn	x		
^{123}Te	x		
^{134}Cs	x		
^{137}Cs	x	x	
^{151}Sm	x		
^{152}Eu	x		
^{192}Ir	x		
^{226}Ra		x	
^{232}U		x	
^{232}Th		x	
^{233}U		x	
^{234}U	x	x	x
^{235}U	x	x	x
^{236}U		x	
^{238}U	x	x	x
^{238}Pu	x	x	
^{239}Pu	x	x	
^{241}Pu	x		
^{241}Am	x	x	
^{242}Pu	x		
^{243}Am	x		
^{244}Cm	x		
^{249}Cf	x		

Information on the concentrations of radioactivity in the wastes is presented in Appendix B. The maximum acceptable concentrations and total radionuclide quantity that can be disposed of at the CWDF will be derived from this pathways analysis study and incorporated into waste acceptance criteria for the facility. The preliminary draft of the waste acceptance criteria is given in Appendix C.

3.2 RADIONUCLIDE CONCENTRATIONS FOR DIRECT INTRUSION PATHWAYS ANALYSIS

To examine a generic scenario that might result from the occupation and use of the CWDF site at some period after it ceases to be controlled by DOE or a replacement institution, a typical waste mass was formulated. The generic scenario is often described as that involving an "inadvertent intruder." Intrusion-type scenarios (e.g., digging into a waste trench to build a house or plant a garden) that involve direct exposure to or ingestion of radioactive materials (except for the water pathway) depend on the maximum concentration of radionuclides at any arbitrary location at the site (see Sect. 2.3). Consequently, conservative (worst-case) source terms for these scenarios are based on the maximum concentration of radionuclides that may be in any specific existing or future waste stream. These concentrations at 100 years after site closure, the assumed time of inadvertent intrusion, are given in Table 3.2. The reduction in radioactivity by decay of specific radionuclides prior to site closure is not taken into account.

3.3 RADIONUCLIDE CONCENTRATIONS FOR WATER PATHWAYS ANALYSIS

A large part, about 7,020 m^3/year (260,000 ft^3/year) of the waste mass will be disposed of in bulk or baled form with little or no containment. The remaining part will essentially consist of fixed sludges (grouted with a concrete mixture). Wastes that are disposed of in bulk or baled form, also referred to as unstabilized wastes (NRC 1982), are of the greatest concern for the water pathways since they are more likely to experience slumping, subsidence, degradation, and therefore much higher infiltration rates and leaching rates than stabilized wastes. The

Table 3.2. Assumed maximum local concentration of radionuclides in disposal units for direct intrusion pathways (100 years after site closure)[a]

Radionuclide	Concentration	
	pCi/cm^3	pCi/g
^{3}H	2.3×10^{2}	1.5×10^{2}
^{14}C	5.5×10^{2}	3.7×10^{2}
^{60}Co	1.0×10^{-3}	6.7×10^{-4}
^{63}Ni	1.3×10^{-4}	8.7×10^{-5}
^{90}Sr	8.8×10^{2}	5.8×10^{2}
^{93}Zr	3.9×10^{2}	2.6×10^{2}
^{99}Tc	2.0×10^{4}	1.3×10^{4}
^{121m}Sn	7.4×10^{1}	4.9×10^{1}
^{123}Te	9.1×10^{-2}	6.6×10^{-2}
^{137}Cs	2.4×10^{3}	1.6×10^{3}
^{151}Sm	2.1×10^{2}	1.4×10^{2}
^{152}Eu	1.6×10^{-3}	1.1×10^{-3}
^{226}Ra	7.7×10^{-1}	5.1×10^{-1}
^{232}U	4.9×10^{-5}	3.3×10^{-5}
^{232}Th	1.9×10^{2}	1.3×10^{2}
^{233}U	5.3	3.5
^{234}U	2.4×10^{1}	1.6×10^{1}
^{235}U	4.0×10^{1}	2.7×10^{1}
^{236}U	1.7×10^{2}	1.1×10^{2}
^{238}U	1.1×10^{3}	7.5×10^{3}
^{238}Pu	5.4	3.6
^{239}Pu	3.9	2.6
^{241}Pu	1.0×10^{-3}	6.7 x 10-4
^{241}Am	1.6×10^{1}	1.1×10^{1}
^{242}Pu	1.3	8.7×10^{-1}
^{243}Am	9.4×10^{-3}	6.3×10^{-3}
^{244}Cm	4.1×10^{-1}	2.9×10^{-7}
^{249}Cf	4.3×10^{-7}	2.9×10^{-7}

[a]Radionuclide decay prior to site closure is not taken into consideration.

characteristics of the unstabilized wastes are, therefore, used to produce conservative source terms for the water pathways analysis. Since the analysis of the water pathways considers that, as a worst-case scenario, leaching of radionuclides from the waste occurs in all disposal units simultaneously, the typical unit waste mass used for the water pathways reflects average concentration of radionuclides in the unstabilized wastes. These concentrations at the time of site closure and at 100 years after closure are given in Table 3.3.

On the basis of the geochemical program (Seeley and Kelmers 1984) and available information on retardation of radionuclides by soil (NRC 1982), it was determined that seven radionuclides (^{90}Sr, ^{99}Tc, ^{137}Cs, ^{3}H, ^{14}C, ^{244}Cm, and ^{238}U) could be selected from Table 3.3 as prototypic of those that may be dispersed through the groundwater pathway. Each radionuclide was selected to conservatively represent a subset of radionuclides on the basis of mass anticipated in the waste, mobility in the soil/groundwater system, and toxicity. These factors were combined to rank the nuclides according to a "hazard rating" (HR), calculated as

$$HR_i = \frac{Q_i}{Rd_i \times MPC_i},$$

where Q_i is the expected content in the waste, Rd_i is the retardation factor, and MPC_i is the maximum permissible concentration of the nuclide i. Maximum permissible concentrations (MPCs) are given in Appendix B of 10 CFR 20. The short-lived nuclides, ^{134}Cs and ^{192}Ir, that have decayed considerably after the first 100 years and the lanthanide, ^{151}Sm, that is notably insoluble are not included in the selection process since they are expected to play only a minor role in the groundwater pathway analysis. Hazard ratings for the radionuclides considered for selection are shown in Table 3.4. Since Rd's generally are a complex function of the concentration of the nuclide in groundwater, average values based on Table 5.2 of NUREG-0945 (NRC 1982) and the results of the geochemical program are listed in Table 3.4. Clearly, the five radionuclides with the highest HRs, ^{90}Sr, ^{99}Tc, ^{137}Cs, ^{3}H, and ^{14}C, need to be studied since they are disposed of in large enough quantities (Table 3.3) that they may limit the permissible activity of other

Table 3.3. Average concentration of radionuclides for water pathways analysis[a]

Radionuclide	Expected activity (Ci/year)	Half-life (year)	Concentrations at time of site closure		Concentrations 100 years after closure	
			Waste (pCi/cm^3)	Leachate (pCi/L)	Waste (pCi/cm^3)	Leachate (pCi/L)
^{3}H	210	1.23×10^{1}	2.98×10^{4}	2.98×10^{7}	1.06×10^{2}	1.06×10^{5}
^{14}C	2	5.73×10^{3}	2.80×10^{2}	2.80×10^{5}	2.80×10^{2}	2.80×10^{5}
^{60}Co	2	5.30	2.80×10^{2}	2.80×10^{5}	5.85×10^{-4}	5.85×10^{-1}
^{90}Sr	34	2.86×10^{1}	4.80×10^{3}	4.80×10^{6}	4.25×10^{2}	4.25×10^{5}
^{93}Zr[b]	2	1.53×10^{6}	2.80×10^{2}	1.63	2.80×10^{2}	1.63
^{99}Tc	4	2.13×10^{5}	5.60×10^{2}	5.60×10^{5}	5.60×10^{2}	5.60×10^{5}
^{121m}Sn[b]	1	5.50×10^{1}	1.40×10^{2}	1.3×10^{2}	3.90×10^{1}	1.3×10^{2}
^{134}Cs	2	2.06	2.80×10^{2}	2.80×10^{5}	$<1.0 \times 10^{-10}$	$<1.0 \times 10^{-7}$
^{137}Cs	82	3.02×10^{1}	1.16×10^{4}	1.16×10^{7}	1.17×10^{3}	1.17×10^{6}
^{151}Sm	2	9.00×10^{1}	2.80×10^{2}	2.80×10^{5}	1.30×10^{2}	1.30×10^{5}
^{192}Ir	4	2.00×10^{-1}	5.60×10^{2}	5.60×10^{5}	$<1.0 \times 10^{-10}$	$<1.0 \times 10^{-7}$
^{234}U	0.01	2.47×10^{5}	1.42	1.42×10^{3}	1.42	1.42×10^{3}
^{235}U[b]	0.23	7.00×10^{8}	3.20×10^{1}	5.66×10^{2}	3.20×10^{1}	5.66×10^{2}
^{238}U[b]	2.73	4.40×10^{9}	3.80×10^{2}	3.48×10^{3}	3.80×10^{2}	3.48×10^{3}
^{238}Pu	0.03	8.78×10^{1}	4.3	4.3×10^{3}	1.95	1.95×10^{3}
^{239}Pu[b]	0.11	2.41×10^{4}	1.56×10^{1}	1.48×10^{1}	1.56×10^{1}	1.48×10^{1}
^{241}Am	0.05	4.32×10^{2}	7.1	7.1×10^{3}	6.05	6.05×10^{3}
^{244}Cm	0.05	1.81×10^{1}	7.1	7.1×10^{3}	1.50×10^{-1}	1.50×10^{2}

[a]Based on average concentrations of radionuclides in the unstabilized wastes. Radionuclide decay prior to site closure is not taken into account.

[b]Solubility limit exceeded in waste.

Table 3.4. Hazard rating of the radionuclides considered in the water pathways

Radionuclide	Retardation factor[a]	Hazard rating	Rank
^{90}Sr	36	3.9×10^{7}	1
^{99}Tc	4	4.6×10^{5}	2
^{137}Cs	350	1.7×10^{5}	3
^{3}H	1	3.5×10^{4}	4
^{14}C	10	3.5×10^{4}	4
^{121m}Sn	1,100	1.2×10^{4}	6
^{238}U	3,520	2.7×10^{3}	7
^{239}Pu	3,520	8.9×10^{2}	8
^{235}U	3,520	3.0×10^{2}	9
^{241}Am	1,200	1.3×10^{2}	10
^{238}Pu	3,520	1.1×10^{2}	11
^{244}Cm	500	4.3×10^{1}	12
^{93}Zr	10,000	3.5×10^{1}	13
^{234}U	3,520	1.3×10^{1}	14
^{60}Co	1,750	6.7×10^{-3}	15

[a]Sources: Based on Table 5.2 in "Nuclear Regulatory Commission, Final Environmental Impact Statement on 10 CFR Part 61, Licensing Requirements for Land Disposal of Radioactive Waste, NUREG-0945, Vol. 1, Office of Nuclear Safety and Safeguards, 1982; and Seeley, F. G., and A. D. Kelmers, Geochemical Information for the West Chestnut Ridge Central Waste Disposal Facility for Low-Level Radioactive Waste, ORNL-6061, Oak Ridge National Laboratory, 1984.

radionuclides to be disposed of. The ^{121m}Sn and ^{60}Co radionuclides (less mobile than ^{137}Cs) can be treated conservatively using the results for the five major elements. The ^{244}Cm nuclide is as toxic as and more mobile than the other transuranic nuclides (^{241}Am, ^{238}Pu, ^{239}Pu) and therefore is conservatively representative of that group. Finally, ^{238}U should be studied as representative of the group of immobile or less mobile radionuclides (^{234}U, ^{235}U, ^{93}Zr).

Thus, ^{90}Sr, ^{99}Tc, ^{137}Cs, ^{3}H, ^{14}C, ^{244}Cm, and ^{238}U are the focus of the groundwater pathway analysis (see Sect. 4). Data for the remaining radionuclides in Table 3.3 are conservatively extrapolated from those for the groundwater pathway.

4. GROUNDWATER PATHWAY FOR LEACHATE (TRENCH DISPOSAL)

As described in Sect. 2.3.2, the groundwater medium provides a potentially significant exposure pathway to man for leachate from radioactive waste disposed of in the study area. A numerical model is used to analyze the migration of contaminants (radionuclides) from the burial trenches through the soil layer and the bedrock aquifers to surface water streams within the study area. The numerical model is first calibrated against existing hydrogeological conditions and verified with transient simulations of observed seasonal variations. The calibrated model is then used to analyze the migration of contaminants under worst-case, yet realistic, conditions. The site hydrodynamic system and the scenarios for the generation of leachate associated with the wetting event are described and analyzed. The migration of contaminants is simulated for a performance period of at least 500 years, and the results are presented for use in the dose analysis (Sect. 6).

4.1 CONCEPTUAL MODEL

To quantify potential radiation exposure to the public and to the inadvertent intruder through the groundwater pathway, the concentration of radioactive materials at two locations is of interest: (1) at a hypothetical water supply point in the Clinch River downstream of the disposal site and (2) at a hypothetical point in the weathered bedrock aquifer located directly underneath a disposal trench. The location of the intruder well in the uppermost tappable water-bearing zone (i.e., weathered bedrock) in the vicinity of the trench is expected to provide the maximum radionuclide concentration and, thus, a conservative estimate of radiation dose to the inadvertent intruder.

A typical two-dimensional vertical cross section is used to simulate the migration of radionuclides from a trench bottom to the intruder well point in the weathered bedrock aquifer. The cross section is located in an area representative of the site conditions, parallel to the expected groundwater flow, and in such a way that it intercepts a large number of trenches (sources), a large number of wells (data points), and collector

areas (discharge locations) as defined by the alluvium of the creeks. The location of the selected cross section is shown in Fig. 4.1. The travel times for migration in the weathered bedrock aquifer and in the collector area are extremely short, as evidenced by the results of the tracer tests and the well's response to storm events (Sect. 2.1.4). Additionally, adsorption characteristics of the rock units were found to be extremely low. These short travel times and low retardation factors in the aquifer will result in what can be considered, for the purposes of the pathways analysis, as instantaneous transport of the radionuclides from the intruder well point to the Clinch River. Because the transport between the intruder well point and the Clinch River is assumed to be instantaneous, the location of the model cross section in relation to the Clinch River is unimportant.

Figure 4.2 shows a schematic design of the conceptual model on the cross section. The cross section is bounded by the alluvium of Grassy Creek to the right and New Zion Creek to the left, which are natural discharge boundaries. The approximate locations of the wells and disposal trenches intercepted by the cross section are indicated. Note that the vertical dimension on Fig. 4.2 is exaggerated by a factor of about 6.

The conceptual model is based on the generalized geologic profile (Fig. 2.3) and consists of a three-layer system that includes a soil layer, a transmissive layer, and a sound bedrock layer.

The upper layer, bounded by the ground surface on top and the top of the weathered bedrock zone on the bottom, represents the soil layer. The materials of the soil layer have relatively low hydraulic conductivities and high adsorption characteristics. Migration of radionuclides in the soil layer is expected to be mainly vertical and associated with time scales that are much longer than those of the other layers. The soil layer constitutes the main buffer layer for radionuclide migration.

Beneath the soil layer is the transmissive layer, a zone characterized by high hydraulic conductivity values and very low adsorption characteristics. The transmissive layer corresponds to the cavitose zone, identified in Sect. 2.1.4, where weathering processes are the most active. Groundwater flows in the cavitose zone are controlled by solution cavities, bedding planes, and discrete joints in the bedrock. Parameters defining the local properties of the materials in the cavitose zone (hydraulic

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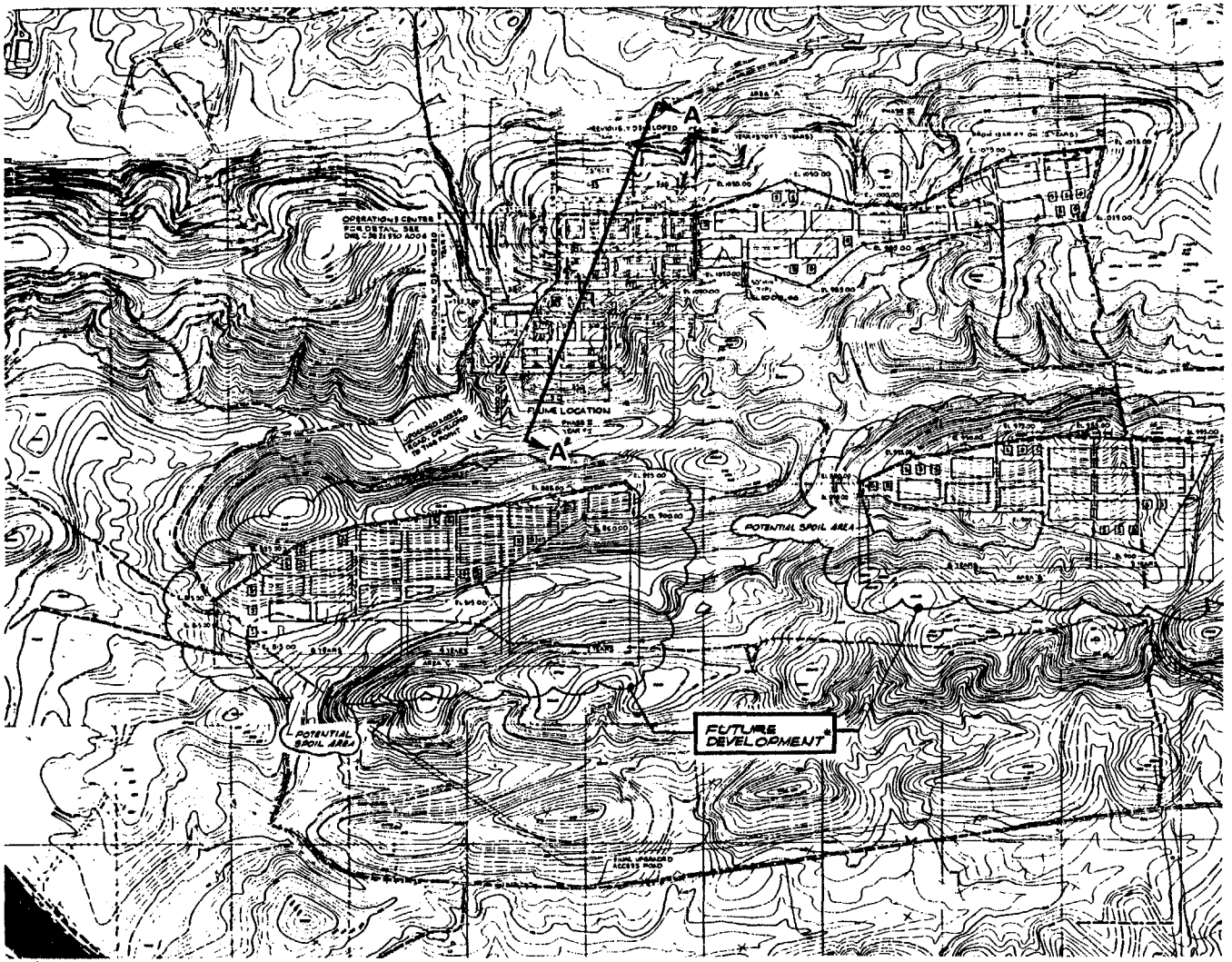

Fig. 4.1. Central Waste Disposal Facility site plot plan showing the planned location of waste disposal trenches and the location of the model cross section.

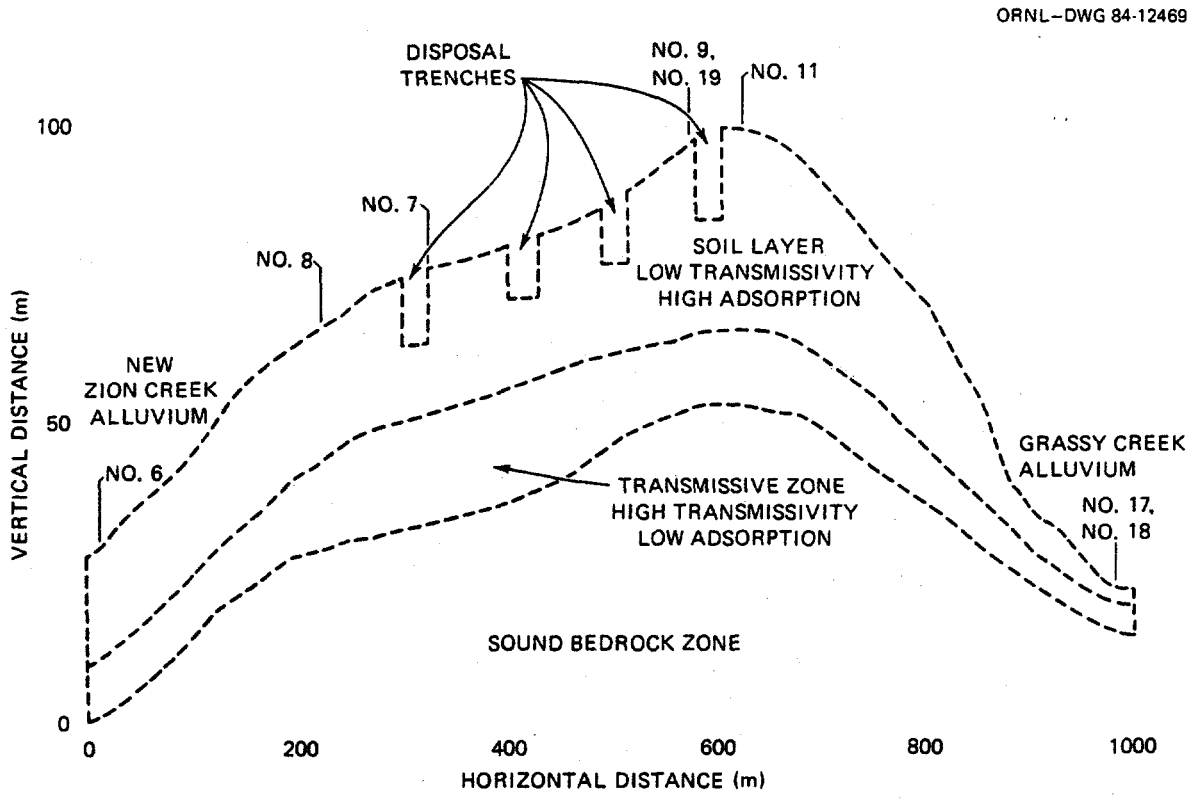

Fig. 4.2. Schematic design of the conceptual model on the cross section.

conductivity, porosity, thickness of the layer, distribution factors, etc.) are expected to vary by orders of magnitude over short distances. Fingering of the migration patterns is expected to occur and to be controlled by fracture orientation. Rapid horizontal transport of contamination fronts in discrete channels and preferential flow paths is also expected. Consequently, the modeling of the transmissive layer is not approached with the general porous media concept. The role of the transmissive layer in the site hydrologic system is interpreted to be the rapid drainage of water that percolates through the soil layer downdip toward the collection or discharge areas. This layer, therefore, acts as a piping system that can be given an overall transmissivity coefficient based on average water table elevations and fluctuations. Because of the extreme complexity and variability of the cavity and channel systems, the vertical extent of the transmisive layer cannot be precisely drawn. It can be approximated for simulation purposes, however, by the extent of the zones of highest transmissivity compared to that of underlying strata. The weathered bedrock layer was reported to be highly transmissive and is, therefore, included in the modeled transmissive layer.

On the basis of packer test results performed in the top of the sound bedrock zone as described by Ketelle and Huff (1984), local or discrete channels and fractures occur in this layer. Although no drilling was performed deeper than 9 m in sound rock, it is expected that the amount of fractures and open joints with high transmissive capacity in the bedrock decreases with depth because of and in relation to the decrease in the weathering activity. Water flow rates in fractures and joints are, therefore, expected to decrease with depth down to an elevation in the bedrock where they become negligible compared to the horizontal flow rates in the upper highly transmissive layers. That elevation in the bedrock is taken as the hypothetical base of the modeled transmissive layer. Below that hypothetical elevation, the sound bedrock is assumed to support flow rates that are negligible compared to the flow rates of maximum impact in the transmissive layer. The dominant parameter defining the transmissive layer is its average transmissivity. Since this parameter will be adjusted on the basis of existing water level conditions in the aquifer, the parameter itself will include the effect of all zones that transmit flow in the weathered and unweathered bedrock zones.

Leachate from the disposal trenches will eventually be transported to the Clinch River. During transit in the groundwater medium and upon mixing in the Clinch River, the concentration of radioactive materials in the leachate will be reduced through dilution. A conservative estimate of the dilution ratio is made as follows. The planned landfill portion of the site covers an area of approximately 7×10^5 m^2 (Ebasco 1984). Assuming that the entire yearly rainfall (1.39 m/year) in a collection area of twice the landfill area infiltrates in the trenches, neglecting evapotranspiration and runoff, a maximum of 1.95×10^9 L/year of water could be contaminated at the site. Dilution of that contaminated water by the Clinch River flow of 150 m^3/s (Boyle et al. 1982) would provide a minimum dilution factor of 2.4×10^3.

4.2 SIMULATION OF MOISTURE MIGRATION AND GROUNDWATER FLOW

This section concerns the simulation of moisture migration and groundwater flow at the site during the preoperational period. The purpose of these simulations is to calibrate a hydrodynamic model of the site against existing hydrogeological conditions, using the data developed during site characterization activities (Ketelle and Huff 1984). The FEMWATER numerical code used for the hydrodynamic transport simulations is described in a report by Yeh and Ward (1980). The code input requirements include the discretization of the model cross section (Fig. 4.2) as a finite element grid, the specification of boundary conditions, and the specification of the parameters describing the soil medium properties (bulk density, porosity, moisture retention capacity, saturated and unsaturated hydraulic conductivities, dispersivity and retardation factor) in each grid cell. Values for these parameters are based on the field and laboratory data presented in the site characterization report (Ketelle and Huff 1984).

4.2.1 Model Grid Layout

The cross section and the finite element grid layout used for the groundwater model are shown in Fig. 4.3. The uppermost grid cells are established to conform with the surface topography along the cross section.

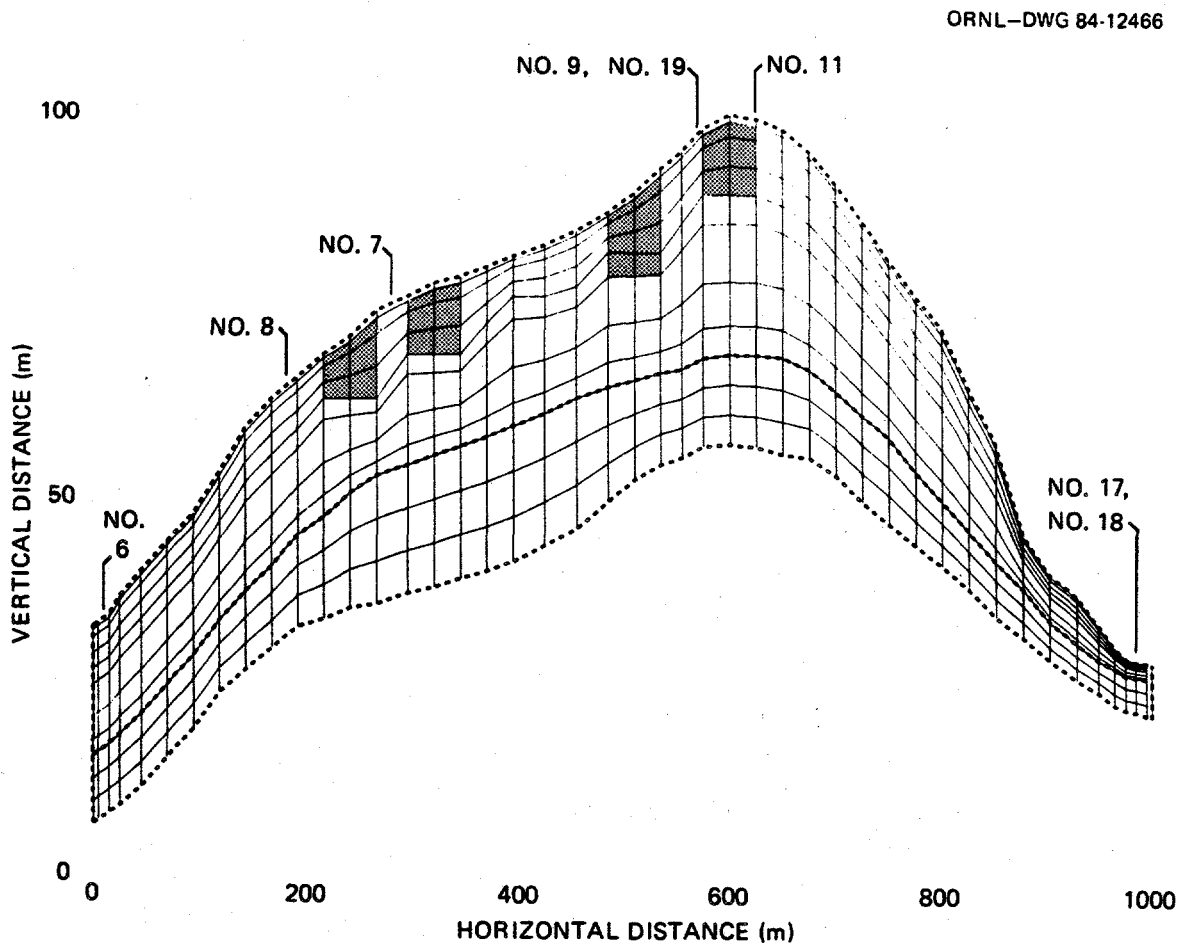

Fig. 4.3. Finite element grid layout on the model cross section.

Specified infiltration fluxes are applied at the top boundary grid cells to simulate infiltration of rainfall. The bottom boundary is specified as a no-flow boundary. This hypothetical boundary corresponds to the elevation of the zone in the sound bedrock below which average vertical water flowrates are not significant compared to the flow rates in the upper layers (see Sect. 2). The side boundaries are selected to correspond to the surface water streams intercepted by the cross section. Surface water streams are an obvious choice for side boundaries since they are natural discharge areas for the local uppermost aquifers that flow under the ridges. Constant head boundary conditions that correspond to groundwater levels at the streams' locations are applied at the side boundaries. The surface separating the soil layer and the weathered bedrock zone is indicated with a heavy dashed line in Fig. 4.3. Also indicated on this figure are the actual locations of existing wells within the cross section and the expected locations of the disposal trenches that are intersected by the cross section. The grid contains 552 node points forming 495 finite elements.

4.2.2 Specification of Model Parameters

Specification of the model parameters is based on two sets of simulations. In the first set, yearly averages of rainfall and water elevations at discharge areas are used in steady state simulations. The soil characteristic parameters are adjusted by trial and error within the range of the field and laboratory data until the results closely correspond with the observed hydrodynamic characteristics of the site. The second set of simulations involves transient simulations of seasonal variations using monthly averaged values of rainfall and water table elevations and verification of the model results against observed water table fluctuations.

The constant head condition on the right-hand side boundary was specified as 244 m to correspond to the recorded groundwater table elevation in the Grassy Creek stream channel. Field observations showed no significant seasonal fluctuations of this elevation which, therefore, can be used for both the steady state and the transient simulations. Similarly, the constant head condition on the left-hand side boundary was specified as 239 m to correspond to the recorded groundwater table

elevation in the New Zion Creek stream channel. This elevation showed slight fluctuations (less than 1 m) during storm events but no seasonal variation about the mean value of 239 m. This latter value was consequently used for the left-hand side boundary condition in the steady state and transient simulations. The infiltration rate at the site varies locally and seasonally with a mean value of 1.52×10^{-6} cm/s. Locally, the infiltration rate varies with ground surface slope. For the steady state simulations, the infiltration rate at the upper boundary was taken to vary from 67% of the mean value for the locations with steeper slopes to 129% for flat ground with a total average over the entire top surface equal to the mean value of 1.52×10^{-6} cm/s. For the transient simulations, the infiltration rate at each cell of the top boundary was modified monthly to conform to the seasonal variations. The ratios of the cell infiltration rate to the mean monthly infiltration rate, corresponding to slope effects, were kept equal to those used in the steady state simulations. The saturated hydraulic conductivity at each grid cell was selected using steady state and transient simulations. In the soil zone, the saturated hydraulic conductivity ranged from 2.0×10^{-5} cm/s to 1.0×10^{-4} cm/s. These values lie within the range of values obtained from the falling-head permeability tests. In the weathered bedrock and transition zone, the saturated hydraulic conductivity ranged from 2.0×10^{-4} cm/s to 1.0×10^{-3} cm/s. These values lie within the range of values obtained from the packer tests and compare favorably with the results of the pump test. As discussed in Sect. 2.1, the laboratory-measured moisture retention curve showing the highest moisture contents versus suction and the associated unsaturated hydraulic conductivity curve were used to simulate the unsaturated characteristics of the soils. In the weathered bedrock and transition zone, the presence of the water table precludes the development of high suction values. Unsaturated flow phenomena, therefore, have little or no effects on the transport of contaminants in that zone and were simulated using the low-suction portion of the average of the laboratory-measured moisture characteristic curves. The effective porosity and the dispersivity were specified as 0.20 and 0 m, respectively, and no anisotropy was included in the model.

4.2.3 Results

The hydrodynamic portion of the model was run for the conditions cited above in steady state and transient simulations. The results of the steady state simulations are shown in Fig. 4.4. The piezometric surface, indicated by the isocurve of zero-pressure head on the figure, compares favorably with the observed mean water table (see Ketelle and Huff 1984). On the top of the ridge, the water table lies within the transition zone, very near the aquitard (bottom boundary). At intermediate elevations, the saturated thickness is large, and the water table is in the upper portion of the weathered bedrock zone. In the New Zion Creek valley, the weathered bedrock zone is fully saturated and the water table lies within the thin weathered bedrock zone. At the monitoring well locations, the simulated water table elevations are within 0.5 m of the observed mean values.

The results of the transient simulations are shown in Figs. 4.5-4.8. On these figures, the simulated water table seasonal fluctuations at various locations on the cross section are indicated and compared with the observed data from monitoring wells. Fig. 4.5 shows that the large seasonal water table fluctuations observed at the ridge top are well simulated by the model. Similarly, Figs. 4.6 and 4.7 show that at low elevations in the valleys, the groundwater elevations observed to be time-invariant are simulated as such by the model. Figure 4.8 shows that the simulated groundwater fluctuations at midhill closely match the observed data.

4.3 SCENARIO DESCRIPTIONS FOR LEACHATE MIGRATION

To quantify the groundwater pathway, several scenarios are defined. The scenarios are not intended to be inclusive of all possible events, but they are expected to represent a conservative, yet realistic, representation of the site under design and off-design conditions. The scenario of the leaching of radionuclides from the waste is considered as the most conservative mode of generation of leachate. Use of this scenario, however, is necessary because of the limited available data on waste form characteristics.

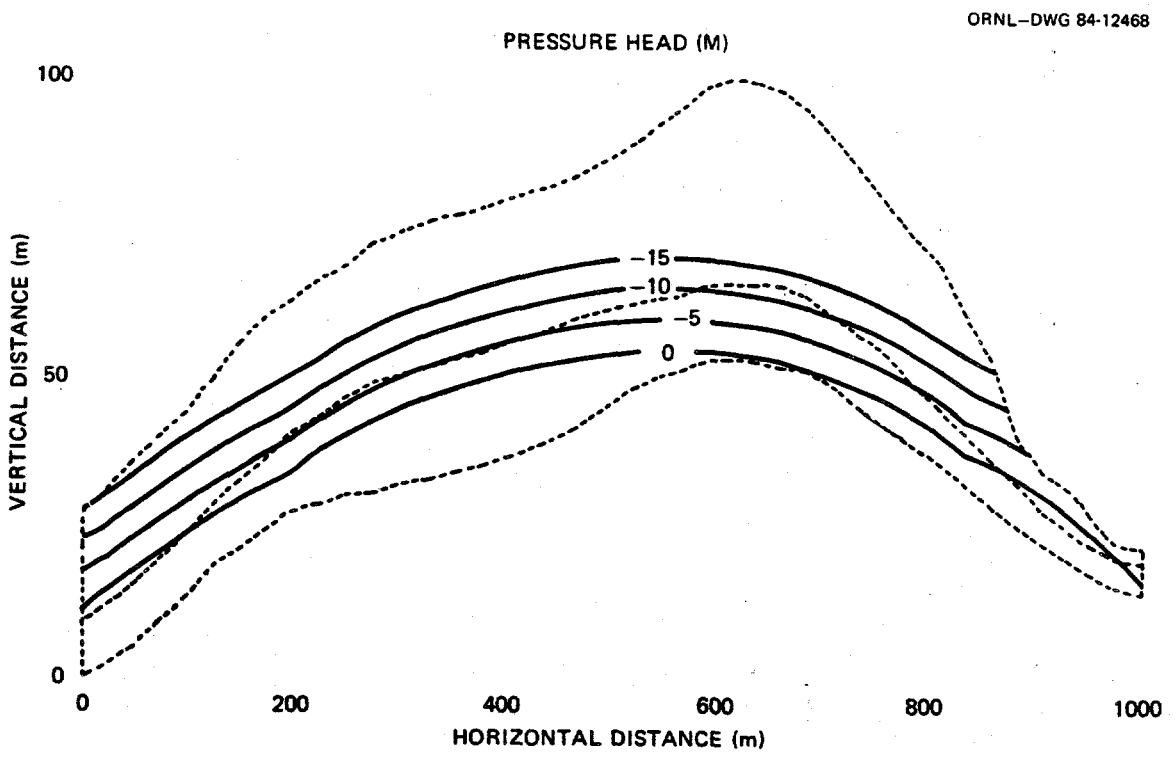

Fig. 4.4. Steady state pattern of pressure head isocurves (the water table is indicated by the curve h=0).

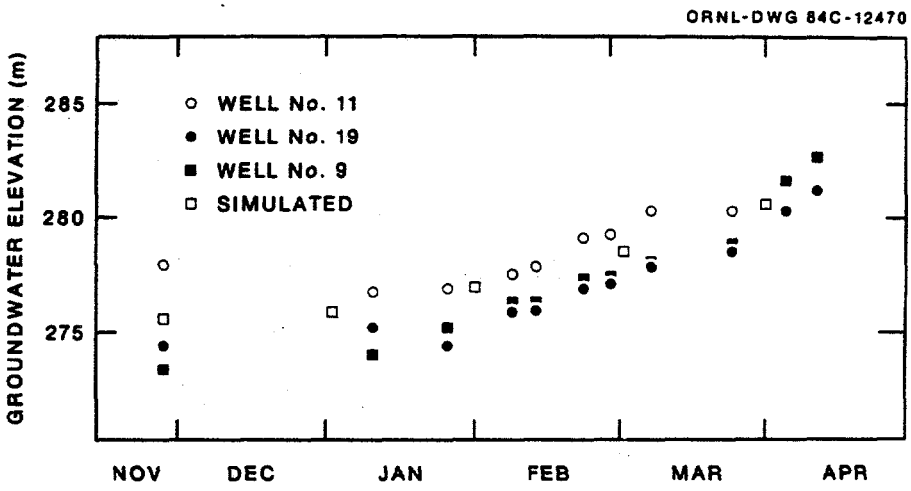

Fig. 4.5. Simulated and observed water table fluctuations for wells located near the top of the ridge.

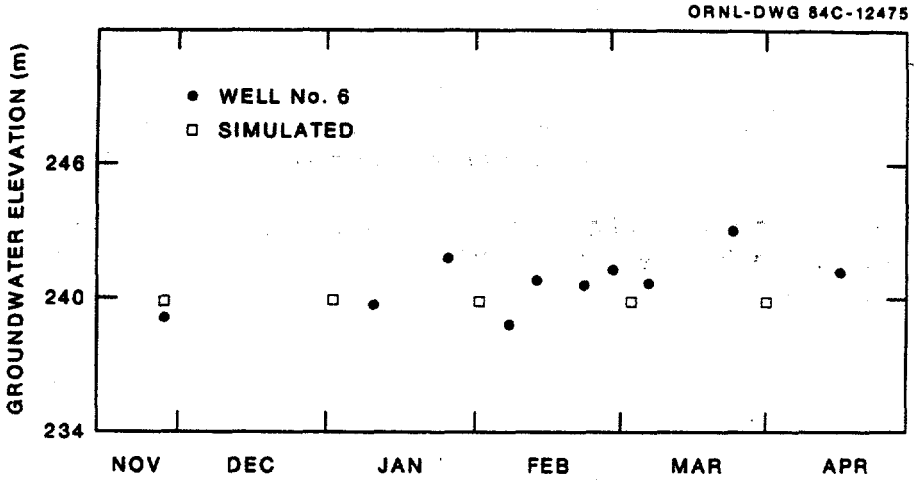

Fig. 4.6. Simulated and observed water table fluctuations for wells located in the New Zion Creek area.

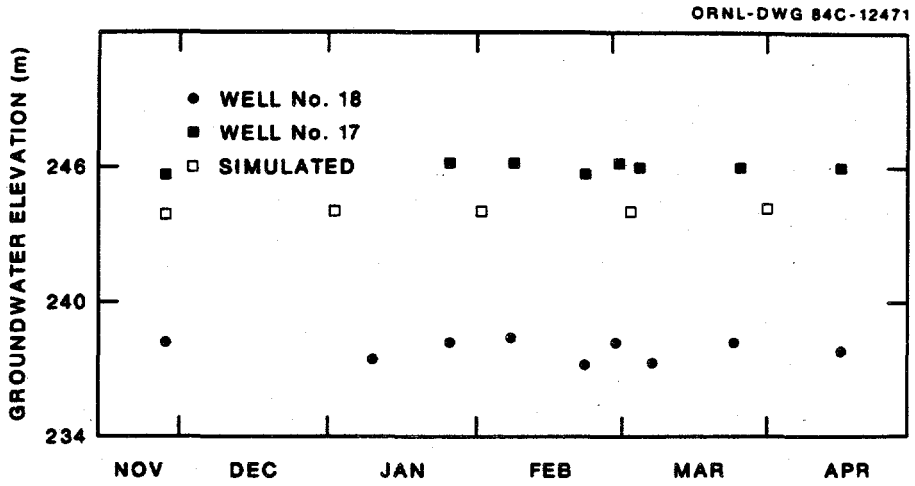

Fig. 4.7. Simulated and observed water table fluctuations for wells located in the Grassy Creek area.

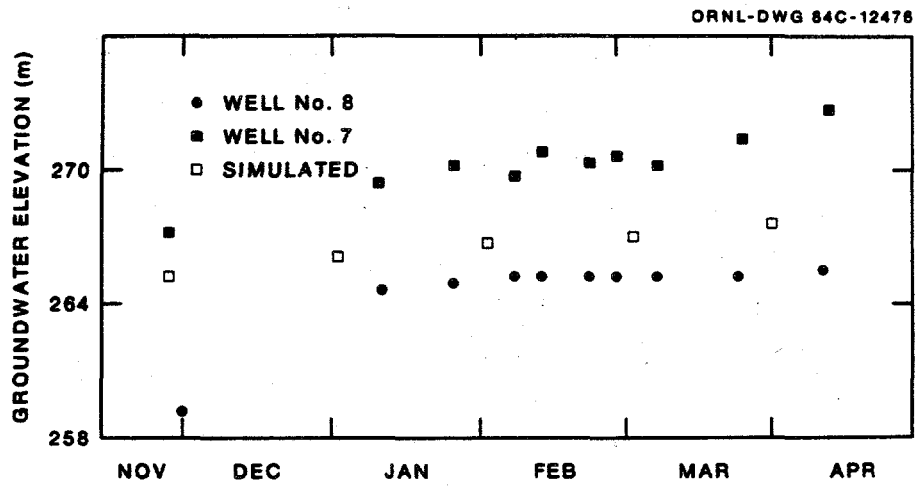

Fig. 4.8. Simulated and observed water table fluctuations for wells located at mid-hill.

Under design conditions, the piezometric surface is always well below the waste disposal units (trenches or tumuli). The caps and the surface drainage systems are designed to prevent the infiltration of surface water into the disposal units. Side wall drains and drainage blankets are designed to prevent the migration of moisture from the undisturbed soil into the trenches by capillary transport. A further discussion of trench design is included in the conceptual design report (Ebasco 1984). During the 100-year period of administrative control, design conditions are expected to prevail and be maintained at the site; that is, the trench caps, the tumulus covers, and the drainage systems are expected to be properly maintained and to operate satisfactorily. Under design conditions, any generation of leachate in the waste disposal units would, therefore, be minimal. Since leachate created in the units would collect in the drainage blanket and be pumped out of the unit if significant accumulation occurs, migration of contamination into the soil would not be significant. Consequently, the analysis of the groundwater pathway assumes that no leachate generation or migration occurs as long as design conditions are maintained at the site. During the institutional control period, however, partial failure of the design conditions could occur and require implementation of remedial actions. The failure is assumed to result in saturation and leaching of part of the waste. Since the probability and mode of failure are uncertain, a scenario (hereafter referred to as the early failure scenario) is defined that conservatively assumed partial failure of all disposal units during the period of institutional controls and 10% migration of the total waste activity into the soils prior to implementation of effective remedial action.

Following the period of institutional control, the integrity of the trench caps or the tumulus covers and the drainage systems could be compromised because of subsidence, cracking, erosion, human intrusion, or other unexpected causes. This damage would result in infiltration of water into the disposal units and saturation of the waste. Following such an event, leachate would be generated and could migrate into the soil. Since maintainance of the site would have ceased, this scenario (hereafter referred to as the postinstitutional failure scenario) assumes that leaching continues until the total mass of radionuclides has been removed from the waste. Scenarios describing the wetting events and the generation

of leachate from the various waste forms for both scenarios were developed using a conservative approach and are described below. The resulting simulation of leachate migration provides predictions of the concentration of leachate in space and time in the soil, groundwater, and surface water systems.

4.3.1 Hydrodynamic Scenario

The hydrodynamic system of the site will be altered under off-design conditions. The infiltration of surface water through the damaged cap rapidly produces standing water at the bottom of the disposal units and leads to the saturation of a portion of the soil and waste mass. Evaporation of the water percolating through the waste mass is minimal. Therefore, the average infiltration rate of contaminated water into the soil at the bottom of the disposal units is taken as 4.4×10^{-6} cm/s, the yearly average rainfall. The location of the disposal units was estimated using data from the conceptual design report (Ebasco 1984). All other input to the hydrodynamic model remained the same as the input used for simulating existing conditions. Figure 4.9 shows the results of the hydrodynamic simulation under off-design conditions. These hydrodynamic conditions are conservatively assumed to prevail during the failure event and the entire postcustodial period and are used as the basis for the simulations of moisture and contaminant migration from the disposal area.

4.3.2 Leaching Scenarios

Two scenarios were developed to describe the generation of leachate from the waste disposed of at the site. Both scenarios assume that leachate is generated during off-design conditions; that is, after failure of the engineered design features (cap, surface water control, side wall drain, and drainage blanket) and, conservatively, that off-design conditions occur simultaneously at all disposal units of the disposal area. The waste is assumed to be disposed of with an equal blend of waste and soil that is com- pacted to a porosity of 0.5. The total waste mass is uniformly distributed among the disposal units, and leaching of the

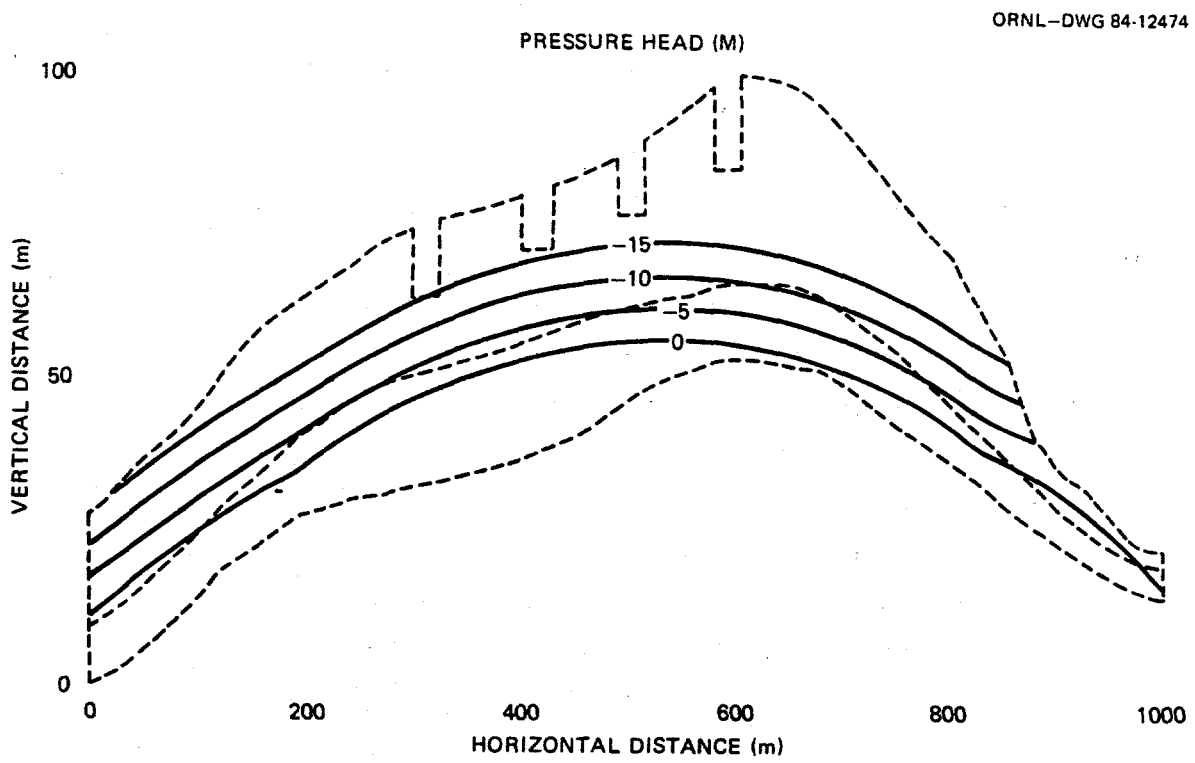

Fig. 4.9. Pattern of pressure head isocurves during the postinstitutional period (the water table is indicated by the curve h=0).

radionuclides occurs in a similar manner in all units. With an infiltration rate of 4.4×10^{-6} cm/s and the assumed porosity of 0.5, total saturation of the waste could occur after 1.6 years. Since no data on leaching characteristics from the waste are available, it is conservatively assumed that the waste is in soluble form and that the radionuclides rapidly dissolve when the waste is saturated with water.

In the early failure scenario, it is assumed that 10% of the total mass of radionuclides is leached from the waste. Since the time and mode of failure of the disposal units are unknown, it is conservatively assumed that failure occurs early during the institutional control period. Consequently, no decay of the waste is assumed, and the concentrations of the radionuclides in the waste are taken as those at site closure (Table 3.3).

In the postinstitutional scenario, leaching is assumed to continue until the total mass of radionuclides has been removed from the waste. Solubility limits in CWDF groundwater have been calculated for some nuclides (see Appendix A) and are used as upper-bound values for concentrations of these nuclides in the leachate. The nuclides for which solubility limits do not apply are assumed to dissolve completely during the first rapid wetting event (1.6 years). The nuclides for which solubility limits apply are assumed to dissolve as controlled by their solubility limits in a series of identical rapid wetting events. Since rapid degradation of the disposal units is expected after institutional controls and maintainance of the site have ceased, the postinstitutional failure scenario is assumed to occur early in the postinstitutional control period. Consequently, the concentrations of radionuclides in the waste are taken as those in the waste at the end of the institutional control period (Table 3.3).

4.4 RESULTS AND DISCUSSIONS

On the basis of the results of the geochemical program and available information on retardation of radionuclides by soil (NRC 1982), the radionuclides of interest for the groundwater pathways analysis have been divided into seven groups (Sect. 3.3). Each group is selected to conservatively represent a subset of radionuclides on the basis of mass

anticipated in the waste, mobility in the soil/groundwater system, and toxicity. ^{3}H, ^{99}Tc, ^{14}C, ^{90}Sr, ^{244}Cm, ^{137}Cs, and ^{238}U, were selected as representative of each group and modeled in the pathways analysis. With this approach, the results for these seven radionuclides are extrapolated to the other radionuclides.

The migration of moisture and radionuclides within the study cross section are simulated using the FEMWASTE computer code (Yeh and Ward 1981). To determine the maximum concentrations of each of the seven radionuclides in the aquifer and in the Clinch River, the analysis is performed for each group, using unit waste concentrations in the disposal units and assuming that the radionuclides do not decay. Maximum dimensionless concentrations in the aquifer (the ratio of the maximum concentration at any point in the aquifer to the leachate concentration) are obtained versus time after the postulated wetting events. The results are then scaled with the appropriate leachate concentration and the appropriate decay constant to provide the maximum concentration of each nuclide that may occur at the intruder well for each scenario (early failure and postinstitutional control failure). Groups 1 through 6, with ^{3}H, ^{99}Tc, ^{14}C, ^{90}Sr, ^{244}Cm, and ^{137}Cs as respective representative radionuclides, use the leaching period of 1.6 years and Kd values of 0, 1, 10, 690, 1,200, and 11,000, respectively. These constant Kd values are conservative values based upon the laboratory batch tests performed as part of the geochemical program (Seeley and Kelmers 1984). The seventh group, with ^{238}U as the representative nuclide, includes radionuclides in which a longer leaching period must be considered because their solubility limits extend the leaching period beyond 1.6 years. The uranium nuclides also exhibit adsorption characteristics that are a strong function of the nuclide concentrations over the range of concentration of interest here. Consequently, each of the uranium nuclides is treated on an individual basis using absolute concentrations and concentration-dependent Kd values and leaching times of 1.6 years, 14.7 years, and 175 years for ^{234}U, ^{235}U, and ^{238}U, respectively.

The results of the numerical simulations are summarized in Figs. 4.10 and 4.11 for Groups 1 through 6; Table 4.1 summarizes the results for

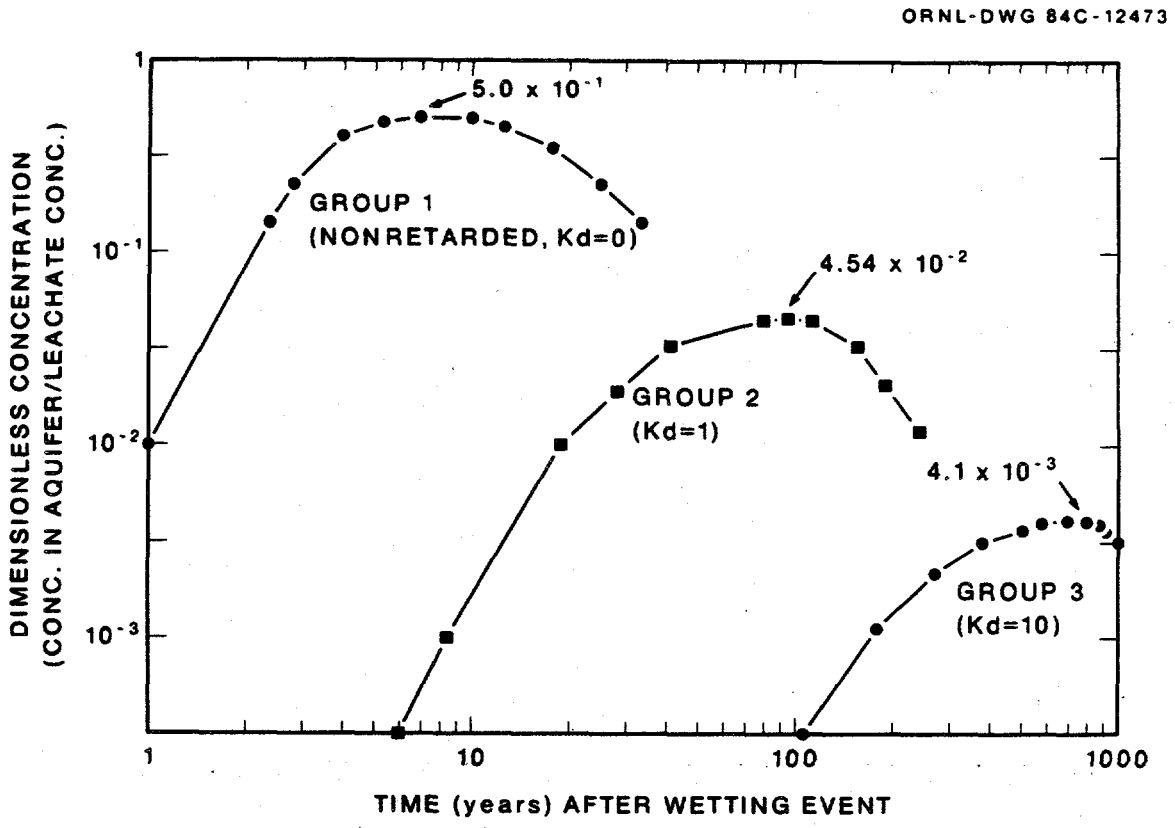

Fig. 4.10. Maximum dimensionless concentrations in the aquifers versus time for Groups 1 through 3, assuming no decay of the nuclides.

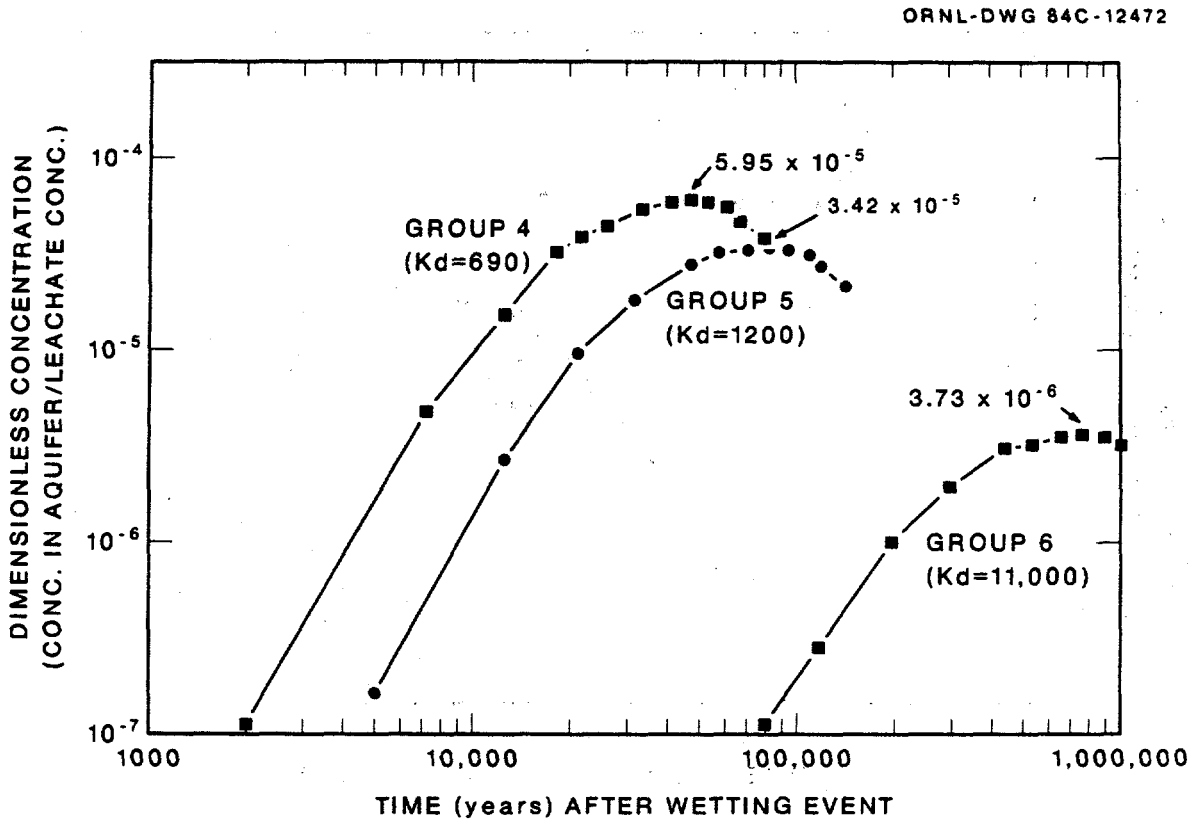

Fig. 4.11. Maximum dimensionless concentrations in the aquifers versus time for Groups 4 through 6, assuming no decay of the nuclides.

Group 7. Figures 4.10 and 4.11 show maximum dimensionless concentrations in the aquifers versus time after the postulated wetting events for Groups 1 through 6, assuming that the nuclides do not decay. The peak value of the nondecayed dimensionless concentration for each group is also indicated. This peak concentration, scaled with the appropriate leachate concentration and decay constant for each nuclide, provides the maximum concentration of each nuclide that may occur at the intruder well. These maximum concentrations are given in Table 4.1 for both the early failure scenario and the postinstitutional failure scenario for the nuclides showing significant concentration in the aquifer (greater than 10^{-10} pCi/L) and are conservatively used in the dose analysis. It is important to note that the peak values for each group of nuclides occur at very different times after the wetting event and that only Groups 1 through 3 reach a maximum concentration in the aquifer within the first 1,000 years. Also important is the fact that, with the exception of ^{3}H, all the short-lived radionuclides decay considerably before reaching the aquifer. This is mainly due to the very high adsorptive capacity of the soil of the West Chestnut Ridge Site, as measured in the geochemical program (Seeley and Kelmers 1984). The approximate period (in years after the wetting event) during which the radionuclide concentration in the aquifer exceeds 50% of the peak value is also indicated in Table 4.1.

Table 4.1. Peak radionuclide concentrations in the aquifers and time of occurrence after wetting event for the early failure scenario and the postinstitutional failure scenario

Radionuclide	Peak concentration in the aquifer (pCi/L)		Time of occurrence[a] (years after wetting event)
	Early failure scenario	Postinstitutional failure scenario	
^{3}H	9.8×10^{5}	3.50×10^{4}	3.5 - 20
^{14}C	1.0×10^{2}	1.05×10^{3}	300 - 1,200
^{93}Zr	1.0×10^{-1}	1.04	>100,000
^{99}Tc	2.5×10^{3}	2.54×10^{4}	30 - 170
^{234}U	6.5	6.45×10^{1}	40 - 200
^{235}U	1.6×10^{1}	1.58×10^{2}	40 - 200
^{238}U	1.6×10^{1}	1.58×10^{2}	40 - 250
^{239}Pu	4.9×10^{-6}	7.13×10^{-2}	30,000 - 150,000
Others		$<10^{-10}$	

[a]Period during which concentration in the aquifer exceeds 50% of the peak concentration.

5. SURFACE WATER PATHWAY FOR LEACHATE (TUMULUS DISPOSAL)

Aboveground disposal in a tumulus is intended to provide containment and isolation of low-level radioactive waste. A description of the tumulus structure is given in Sect. 2.2.2. The facility has the potential to perform without the release of radioactivity throughout the performance period; however, complete containment and isolation cannot be assured. Since experience with the tumulus disposal concept is limited, the loss of containment or isolation is considered as the basis of the pathways analysis. This section provides an analysis of the transport of radioactivity through the water pathway, which is used in the radiological dose analysis (Sect. 6).

5.1 SCENARIOS FOR LEACHATE GENERATION

Two types of failure are considered for analyzing the water pathway associated with aboveground disposal with the tumulus concept. In the first scenario, the failure of the design occurs during the administrative control period immediately following site closure. The second scenario considers failure of the tumulus as a result of intrusion by human or natural processes that could permit an uncontrolled release of radioactivity. These two scenarios are described in the following sections.

5.1.1 Institutional Control

The probability and mode of failure of the design for an aboveground disposal facility using the tumulus concept are unknown. The tumulus concept is intended to reduce to the minimum the probability of occurrence of the anticipated modes of failure. However, unknown site characteristics or unanticipated events during facility construction and operation could result in failure of the concept. Some possible causes of failure that can be envisioned include faulty construction, land subsidence, sinkhole formation, erosion of the trench cover, and clogging of the drainage system

Faulty construction of the disposal facility could compromise the integrity of the tumulus through flaws such as a cracked cement base,

improper grading of the interior floor drains, poor external drainage, and cracks and fissures in the layered cap. Although some land subsidence is expected as a result of decomposition and consolidation of the waste, excessive subsidence that could occur from unstable waste stacking or improper backfilling would compromise the isolation of the waste. Sinkhole formation from karst activity is not anticipated to occur in the areas to be used for waste disposal, even though these formations are typically observed in the Knox Group in the site area. If a sinkhole were to develop beneath one or more tumuli, the containment and isolation of the waste would be breached. Excessive erosion of the layered cap could occur as a result of poor site maintenance or slumping of the steep sides of the tumulus. The internal or external drainage system could become clogged from erosion or excessive infiltration into the waste mass.

All of the modes of failure discussed above have the potential for premature saturation of the waste and the subsequent formation of transportable leachate. Since the tumulus is above the natural grade, the leachate could be discharged directly to the surface and migrate overland to surface water. This type of failure would result in the most rapid transport of leachate to an individual or the public. The mode of failure that would generate the greatest impact would be a failure occurring in all of the tumuli at the site. This type of failure would be attributable to a generic design or construction defect that would not be easily detected. In what could be considered to be the more likely type of failure, that of an individual tumulus (such as the formation of a sinkhole beneath a tumulus), the resulting concentrations of radioactivity in the surface water would not be as large as those resulting from failure of all the tumuli.

The scenario used to analyze the failure of the tumulus concept assumes that all the tumuli partially fail simultaneously during the institutional control period and the leachate that is generated is discharged to the surface across the site area. Ten tumuli are assumed to contribute leachate to the Station 3 monitoring location, 20 tumuli are assumed to contribute leachate to the Station 2 monitoring location, and 30 tumuli are assumed to contribute leachate to the Station 1 monitoring location. The remaining tumuli are considered to be outside the Ish Creek watershed and, therefore, would not contribute to the contamination of Ish Creek. The locations of Stations 1, 2, and 3 are shown in Fig. 2.10.

A failed tumulus is assumed to discharge leachate with a flux equivalent to 10% of the incident precipitation contacting the waste. Of the average annual precipitation of 140 cm/year (55 in./year), 50% is assumed to contact the waste, and the remaining precipitation is assumed to become runoff, evapotranspiration, or infiltration. The leachate flux discharged to Ish Creek is then calculated to be 6.99 cm/year (2.75 in./year) of water distributed over the 1,900-m^2 (20,000-ft^2) area associated with each tumulus or 130 m^3/year (4,580 ft^3/year) per tumulus, which is equivalent to 4.2×10^{-6} m^3/s (1.5×10^{-4} ft^3/s). The initial concentration of the leachate is assumed to be the leachate concentration shown in Table 3.3 for design failures that occur at the time of site closure or at 100 years after site closure.

During the institutional control period, access to the site by an inadvertent intruder would be unlikely. Potential exposure from the failure of the tumulus design would be possible only at the edge of the buffer zone or site boundary. Stations 1, 2, and 3 are analyzed such that these results can be used as an aid in identifying the necessary extent of the buffer zone.

The surface water discharge to Ish Creek is assumed to be well represented by the data collected between July 15, 1983, and July 11, 1984. These data are summarized in Tables 2.4 and 2.5. From these data, the dilution factors for leachate discharged from the tumuli are calculated as.

$$\text{Dilution factor} = \frac{\text{(mean annual flow of Ish Creek)} + [\text{(leachate discharge/tumulus)} \times \text{(number of tumuli)}]}{\text{(leachate discharge/tumulus)} \times \text{(number of tumuli)}}.$$

The dilution factors are presented in Table 5.1. The discharge of the Clinch River at Melton Hill Dam averages 150 m^3/s (5,280 ft^3/s). Since all 60 tumuli would ultimately discharge leachate to the Clinch River, the dilution factor is calculated as:

$$\text{Dilution factor} = \frac{\text{Clinch River flow}}{\text{(leachate discharge/tumulus)} \times \text{(number of tumuli)}}.$$

The dilution factor is 5.9×10^5.

5.1.2 Postinstitutional Control

Following the end of institutional control, individual exposure can result from an inadvertent intruder entering the site area (Sect. 2.2.1). Public exposure can result from contamination of a drinking water source. The Clinch River is the nearest and only potential public water source. The intruder, however, could obtain drinking water and domestic water either from the Clinch River or from Ish Creek.

During the institutional control period, any leachate that is generated would be drained by the internal drainage system of the tumulus, providing the tumulus performs as designed. The leachate would be collected in sumps, removed if any contamination were present, treated, and returned to the tumulus. Even though leachate discharge could be reduced to minimal levels during institutional control, the concrete floor would not prevent moisture from accumulating within the tumulus but would only impede the transport of contaminants. Since the tumulus would be open during disposal operations and upper soil horizons could become saturated after heavy rainfall, the waste packages would be moist at closure and remain moist throughout much of the institutional control period. The moisture present in the tumulus would promote the degradation of the waste packages during institutional control, and the radionuclides within the wastes would become available for transport at the end of institutional control. For analyzing the worst case for postinstitutional control activities, all of the radionculides within the tumulus are assumed to be retained throughout institutional control.

Following the end of institutional control, the infiltration-resistant cover could be rendered ineffective and allow precipitation and runoff to enter the tumulus. The cover could be rendered ineffective by water erosion, subsidence, or earth-moving equipment. Since the tumulus is above-ground and has a concrete floor and an installed drainage system, the waste is not likely to be inundated by water as in the case of shallow land burial. As a result, the leaching period is assumed to be extended, the leachate is assumed to be discharged directly to surface water, and the surface water is assumed to be used as a drinking water supply by an inadvertent intruder.

Table 5.1. Dilution factors for institutional control period analysis of Ish creek

Station	Dilution factor
1	305
2	260
3	320

The initial concentrations of radionuclides in the tumulus-generated leachate at the end of institutional control are shown in Table 3.3. The concentration of the leachate is assumed to decay exponentially in time with 90% of the activity leached at 50 years after the onset of leachate transport.

The scenario used for analyzing failure of the tumulus in the post-institutional control period assumes that all the tumuli fail simultaneously 100 years after site closure. Ten tumuli are assumed to contribute leachate to the Station 3 monitoring location, 20 tumuli are assumed to contribute leachate to the Station 2 monitoring location, and 30 tumuli are assumed to contribute leachate to the Station 1 monitoring location. The remaining tumuli are considered to be outside the Ish Creek watershed. The locations of Stations 1, 2, and 3 are shown in Fig. 2.10. The leachate flux is assumed to be 50% of the incident annual precipitation to give credit to the effects of overland runoff, evapotranspiration, and infiltration. Since the annual average precipitation is 140 cm (55 in.), the leachate flux is then 6.99 cm/year (27.5 in./year) of water distributed over 1,900 m^2 (20,000 ft^2) per tumulus or 1,300 m^3/year (45,800 ft^3/year), which is equivalent to 4.2×10^{-5} m^3/s (1.5×10^{-3} ft^3/s).

The dilution factors for Ish Creek and the Clinch River are calculated using the same method described in Sect. 5.1.1. The resulting dilution factors for Ish Creek are shown in Table 5.2. The dilution factor for the Clinch River is 5.9×10^4.

5.2 ANALYSIS

The scenarios for analyzing the tumulus water pathway during both institutional control and postinstitutional control are by surface water migration of leachate. Groundwater migration of leachate also could occur, but exposure from a combined groundwater and surface water scenario would be less than the surface water scenario because adsorption and decay in soil would reduce the concentrations of radioactivity in the groundwater. Since surface waters have limited adsorptive capacity by comparison with that of groundwater and the transport of leachate in surface waters is rapid, the potential exposure from leachate migration is maximized.

Table 5.2. Dilution factors for postinstitutional control period analysis of Ish Creek

Station	Dilution factor
1	31
2	27
3	33

The normalized concentrations of the radionuclides discharged from the failed tumulus as a function of time for both scenarios are shown in Fig. 5.1. As can be seen from this figure, the peak concentration occurs immediately following the failure of the tumulus and decreases with time. The discharged leachate is then subject to dilution by the natural flows of Ish Creek and the Clinch River.

5.2.1 Institutional Control

The normalized concentration of radionuclides in Ish Creek decreases with time, as shown in Fig. 5.2. The reduction in concentration is attributable to the dilution of the leachate by the natural flows of Ish Creek. The peak concentrations of radionuclides at Stations 1, 2, and 3 and the Clinch River are listed in Table 5.3. The peak concentrations shown in Table 5.3 are based on the assumption that decay of the waste activity had not occurred prior to the failure of the tumulus and that the failure occurred immediately after site closure. If the failure had occurred later in the institutional control period, the peak concentrations would be reduced as a result of radioactive decay. Since Stations 1, 2, and 3 are within the buffer zone and site boundary for the waste disposal facility, the possibility that an inadvertent intruder would be exposed to these concentrations of radioactivity is unlikely. If the tumulus were to fail 100 years after site closure, which corresponds to the institutional control period, the peak concentrations of radionuclides at Stations 1, 2, and 3 and the Clinch River would be those listed in Table 5.4.

5.2.2 Postinstitutional Control

Potentially, the inadvertent intruder could establish a water supply at Stations 1, 2, and 3 following the end of institutional control. Above Station 3 on Ish Creek, however, the flows are too low for a drinking water supply even with storage. The peak concentration of radionuclides in Ish Creek at Stations 1, 2, and 3 and the Clinch River for the scenario described in Sect. 5.1.2 are presented in Table 5.5. The normalized concentration of radionuclides in Ish Creek decrease with time as shown in

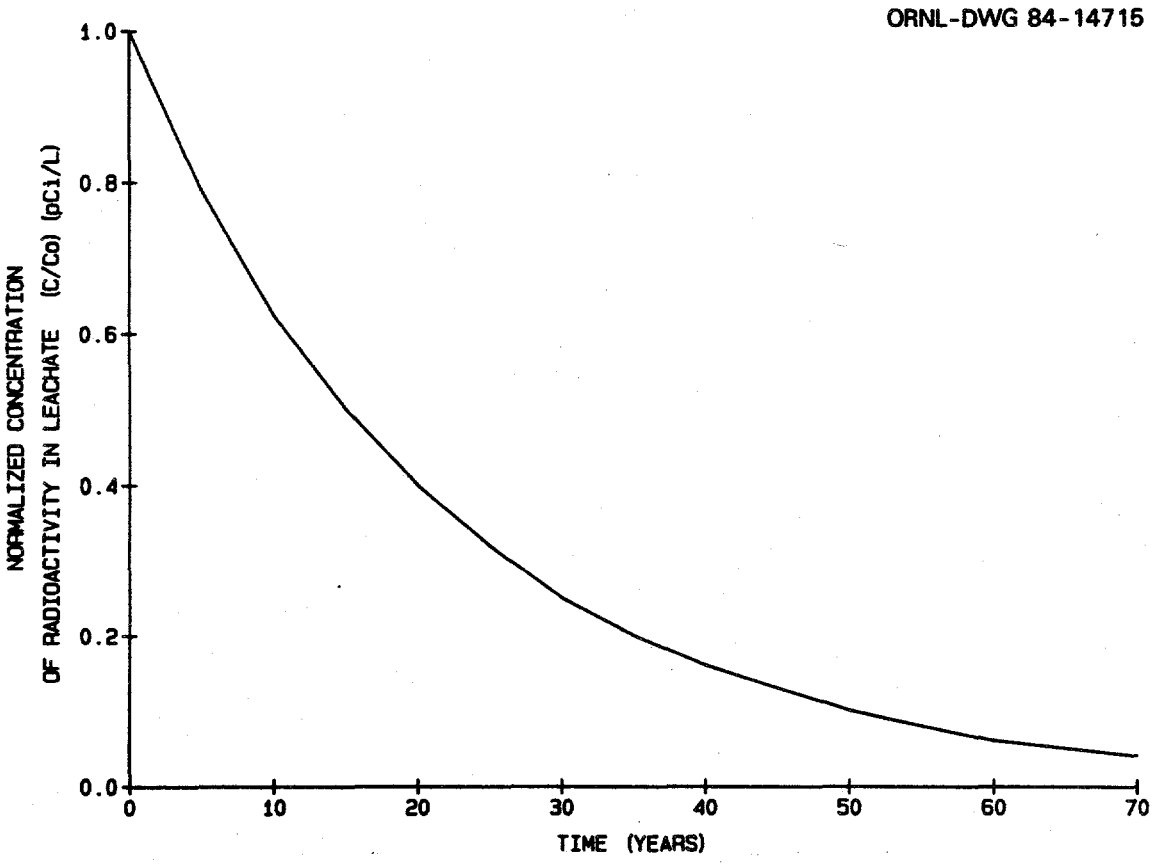

Fig. 5.1. Assumed normalized concentration of radioactivity in leachates discharged from the aboveground disposal alternative.

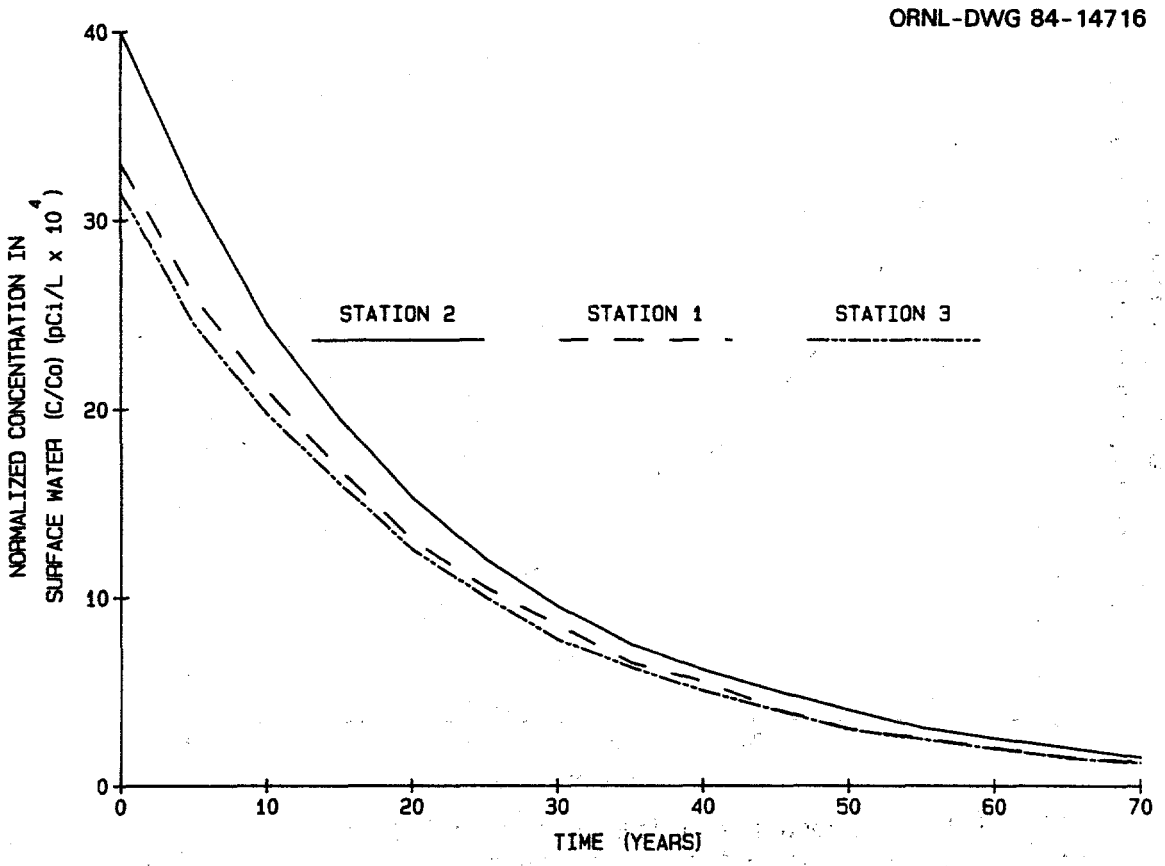

Fig. 5.2. Normalized concentration of radioactivity at Stations 1, 2, and 3 and in Ish Creek for the institutional control period after the release of radionuclides from the tumuli.

Table 5.3. Peak concentration of radionuclides in surface waters from early failure of tumulus during institutional control[a] (Units are pCi/L)

Radionuclide	Ish Creek			Clinch River
	Station 1	Station 2	Station 3	
^{3}H	9.77×10^{4}	1.15×10^{5}	9.31×10^{4}	5.05×10^{1}
^{14}C	9.18×10^{2}	1.08×10^{3}	8.74×10^{2}	4.75×10^{-1}
^{60}Co	9.18×10^{2}	1.08×10^{3}	8.75×10^{2}	4.75×10^{-1}
^{90}Sr	1.57×10^{4}	1.85×10^{4}	1.50×10^{4}	8.14
^{93}Zr	5.34×10^{-3}	6.27×10^{-3}	5.09×10^{-3}	2.76×10^{-6}
^{99}Tc	1.84×10^{3}	2.15×10^{3}	1.75×10^{3}	9.49×10^{-1}
^{121m}Sn	4.26×10^{-1}	4.99×10^{-1}	4.06×10^{-1}	2.21×10^{-4}
^{134}Cs	9.18×10^{2}	1.08×10^{3}	8.75×10^{2}	4.75×10^{-1}
^{137}Cs	3.80×10^{4}	4.46×10^{4}	3.63×10^{4}	1.97×10^{1}
^{151}Sm	9.18×10^{2}	1.08×10^{3}	8.75×10^{2}	4.75×10^{-1}
^{192}Ir	1.84×10^{3}	2.15×10^{3}	1.75×10^{3}	9.49×10^{-1}
^{234}U	4.66	5.46	4.44	2.41×10^{-3}
^{235}U	1.86	2.18	1.77	9.59×10^{-4}
^{238}U	1.14×10^{1}	1.34×10^{1}	1.09×10^{1}	5.90×10^{-3}
^{238}Pu	1.33×10^{1}	1.57×10^{1}	1.27×10^{1}	7.29×10^{-3}
^{239}Pu	4.85×10^{-2}	5.69×10^{-2}	4.63×10^{-2}	2.51×10^{-5}
^{241}Am	2.33×10^{1}	2.73×10^{1}	2.22×10^{1}	1.20
^{244}Cm	2.33×10^{1}	2.73×10^{1}	2.22×10^{1}	1.20×10^{-2}

[a]Assumes that design failure occurs at the time of site closure. Decay of radionuclides has not been taken into account.

Table 5.4. Peak concentration of radionuclides in surface waters from failure of tumulus at end of institutional control period[a] (Units are pCi/L)

Radionuclide	Ish Creek			Clinch River
	Station 1	Station 2	Station 3	
^{3}H	3.48×10^{2}	4.08×10^{2}	3.31×10^{2}	1.80×10^{-1}
^{14}C	9.18×10^{2}	1.08×10^{3}	8.75×10^{2}	4.75×10^{-1}
^{60}Co	1.92×10^{-3}	2.25×10^{-3}	1.83×10^{-3}	9.92×10^{-7}
^{90}Sr	1.39×10^{3}	1.63×10^{3}	1.33×10^{3}	7.20×10^{-4}
^{93}Zr	5.34×10^{-3}	6.27×10^{-3}	5.09×10^{-3}	2.76×10^{-6}
^{99}Tc	1.84×10^{3}	2.15×10^{3}	1.75×10^{3}	9.49×10^{-1}
^{121m}Sn	4.27×10^{-1}	5.01×10^{-1}	4.07×10^{-1}	2.21×10^{-4}
^{134}Cs	$<10^{-13}$	$<10^{-13}$	$<10^{-13}$	$<10^{-15}$
^{137}Cs	3.84×10^{3}	4.50×10^{3}	3.66×10^{3}	1.98
^{151}Sm	4.26×10^{2}	5.00×10^{2}	4.06×10^{2}	2.20×10^{-1}
^{192}Ir	$<10^{-13}$	$<10^{-13}$	$<10^{-13}$	$<10^{-15}$
^{234}U	4.66	5.46	4.44	2.41×10^{-3}
^{235}U	1.86	2.18	1.77	9.59×10^{-4}
^{238}U	1.14×10^{1}	1.34×10^{1}	1.09×10^{1}	5.90×10^{-3}
^{238}Pu	6.39	7.50	6.09	3.31×10^{-3}
^{239}Pu	4.85×10^{-2}	5.69×10^{-2}	4.63×10^{-2}	2.51×10^{-5}
^{241}Am	1.98×10^{1}	2.33×10^{1}	1.89×10^{1}	1.03×10^{-2}
^{244}Cm	4.92×10^{-1}	5.77×10^{-1}	4.69×10^{-1}	2.54×10^{-4}

[a]Assumes that design failure occurs 100 years after site closure. The concentrations represent the values of Table 5.3 reduced by radioactive decay.

Table 5.5. Peak concentration of radionuclides in surface waters from tumulus failure during postinstitutional control period (Units are pCi/L)

Radionuclide	Ish Creek			Clinch River
	Station 1	Station 2	Station 3	
^{3}H	3.42×10^{3}	3.93×10^{3}	3.21×10^{3}	1.80
^{14}C	9.03×10^{3}	1.04×10^{4}	8.48×10^{3}	4.75
^{60}Co	1.89×10^{-2}	2.17×10^{-2}	1.77×10^{-2}	9.92×10^{-6}
^{90}Sr	1.37×10^{4}	1.57×10^{4}	1.29×10^{4}	7.20×10^{-3}
^{93}Zr	5.26×10^{-2}	6.04×10^{-2}	4.94×10^{-2}	2.76×10^{-5}
^{99}Tc	1.84×10^{4}	2.07×10^{4}	1.70×10^{4}	9.49
^{121m}Sn	4.19	4.82	3.94	2.21×10^{-3}
^{137}Cs	3.77×10^{4}	4.33×10^{4}	3.55×10^{4}	1.98×10^{1}
^{234}U	4.58×10^{1}	5.26×10^{1}	4.30×10^{1}	2.41×10^{-2}
^{235}U	1.83×10^{1}	2.10×10^{1}	1.72×10^{1}	9.59×10^{-3}
^{238}U	1.12×10^{2}	1.29×10^{2}	1.05×10^{2}	5.90×10^{-2}
^{238}Pu	6.29×10^{1}	7.22×10^{1}	5.91×10^{1}	3.31×10^{-2}
^{239}Pu	4.77×10^{-1}	5.48×10^{-1}	4.48×10^{-1}	2.51×10^{-4}
^{241}Am	1.95×10^{2}	2.44×10^{2}	1.83×10^{2}	1.03×10^{-1}
^{244}Cm	4.84	5.56	4.55	2.54×10^{-3}

Fig. 5.3. Likewise, the decrease of the normalized concentration of radionuclides in the Clinch River is shown in Fig. 5.4. The peak concentrations for the postinstitutional control period are greater than the peak concentrations for the institutional control period because of the complete loss of containment and isolation included in the postinstitutional control scenario.

5.3 SUMMARY

The potential migration of radioactivity in the water pathway from the tumulus alternative has been determined for conditions of design failure and from occupancy of the site and penetration of waste material by an inadvertent intruder. Failure of the design is considered to occur during the institutional control period and result in partial failure of the tumulus in containment and isolation of the waste. Inadvertent intrusion is considered to result in complete failure of containment and isolation of the waste. The radioactivity migrating from the waste is considerd to be released to surface water as a conservative analysis of the potential transport of radionuclides. The doses receivable from these exposures are discussed in Sect. 6.

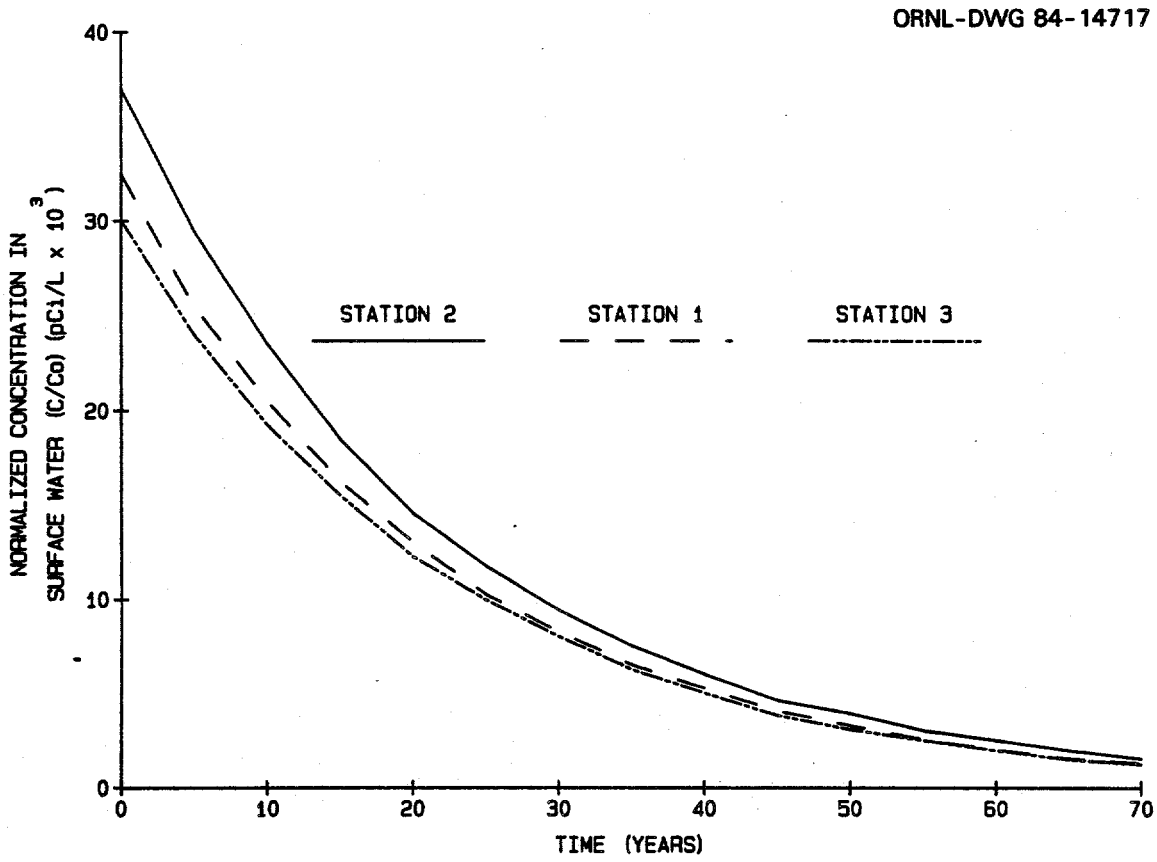

Fig. 5.3. Normalized concentration of radioactivity at Stations 1, 2, and 3 and in Ish Creek for the postinstitutional control period after the release of radionuclides from the tumuli.

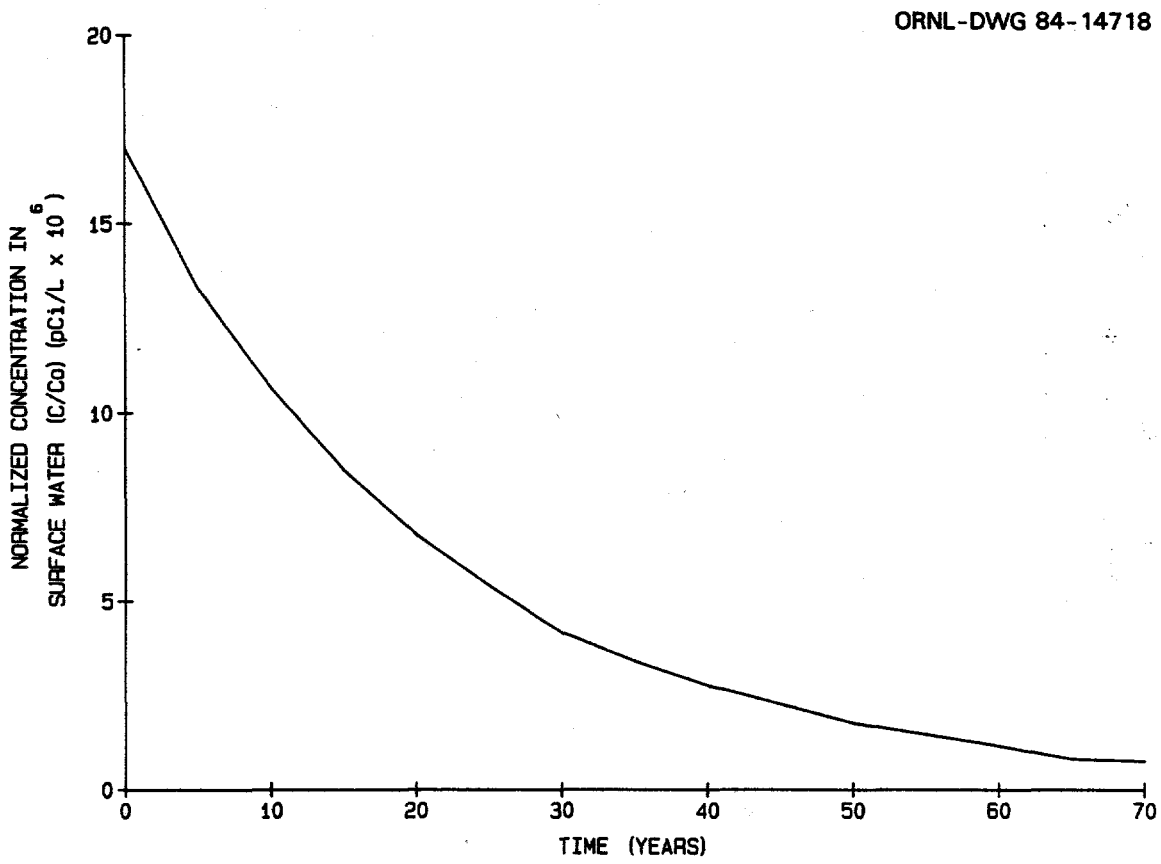

Fig. 5.4. Normalized concentration of radioactivity in the Clinch River for public exposure from the tumulus postinstitutional control period.

6. RADIOLOGICAL DOSE ANALYSIS

This section provides radiation dose estimates for the potentially viable exposure pathways discussed in Sect. 2.3. The concentrations of radionuclides in water that could be ingested by an inadvertent intruder and individuals outside the site boundary are given in Sect. 4 for trench disposal and Sect. 5 for tumulus disposal. The calculated dose commitments are compared to the limits provided in DOE Order 5480.1A for the inadvertent intruder and to 40 CFR 190 and 40 CFR 191 for individuals outside the site boundary.

6.1 METHODOLOGY

Both external and internal doses could result from radioactive waste disposal at the CWDF site. In this study, doses from external exposure are annual doses, while the doses from internal exposures to inhaled and ingested radionuclides are 50-year dose commitments--the estimate of the total dose an individual will receive from 1 year of radionuclide intake, integrated over the next 50 years of his life.

The methodology for making estimates of radiation dose following the release of radionuclides to the environment has been presented in outline form and selected detail in NRC and Oak Ridge National Laboratory (ORNL) reports (Adams and Rogers 1978; Killough and McKay 1976).

The dose conversion factors for estimating doses from the intake of radionuclides through inhalation and ingestion are available in a report by Dunning et al. (1981) and those for estimating doses from external radiation are in a report by Kocher (1981). Dose conversion factors pertinent to this assessment are given in Tables 6.1, 6.2, and 6.3. Sample problems for internal dose calculations are shown on pp. 4-108 and 4-109 of ORNL-4992 (Killough and McKay 1976), on pp. 42 and 43 of ORNL/OEPA-7 (Hill 1979), and in Appendix B of ORNL-5529 (Miller et al. 1980).

The environmental parameters used in estimating doses are given in Regulatory Guide 1.109 (NRC 1977). Many of the basic parameters are conservative; that is, where site-specific information is unknown, the values are chosen to maximize human intake or exposure. In estimating the dose via ingestion of vegetables and water, it is assumed that 10% of the

Table 6.1. Dose conversion factors for ingestion doses (rem/μCi)

Radionuclide	Organ			
	Total body	Bone	Kidney	Lungs
^{3}H	8.3×10^{-5}	3.7×10^{-5}	8.5×10^{-5}	8.4×10^{-5}
^{14}C	1.9×10^{-3}	1.2×10^{-3}	1.1×10^{-3}	8.5×10^{-4}
^{60}Co	4.4×10^{-3}	3.8×10^{-3}	5.7×10^{-3}	8.6×10^{-3}
^{63}Ni	4.8×10^{-5}	9.2×10^{-4}	1.1×10^{-4}	1.6×10^{-4}
^{90}Sr	9.4×10^{-2}	1.2	6.0×10^{-3}	5.9×10^{-3}
^{93}Zr	1.0×10^{-6}	3.8×10^{-5}	9.7×10^{-6}	9.5×10^{-6}
^{99}Tc	2.1×10^{-4}	3.6×10^{-4}	4.6×10^{-4}	3.2×10^{-4}
^{121m}Sn	7.4×10^{-4}	3.0×10^{-2}	5.0×10^{-4}	7.4×10^{-3}
^{123}Te	7.6×10^{-4}	2.4×10^{-3}	1.1×10^{-2}	2.6×10^{-3}
^{137}Cs	4.9×10^{-2}	6.8×10^{-2}	7.7×10^{-2}	1.0×10^{-1}
^{151}Sm	2.8×10^{-6}	6.9×10^{-5}	1.3×10^{-5}	1.0×10^{-5}
^{152}Eu	3.9×10^{-5}	1.9×10^{-4}	2.1×10^{-4}	1.0×10^{-4}
^{226}Ra	3.4	4.3×10^{1}	5.9×10^{-1}	5.9×10^{-1}
^{232}U	2.1	2.9×10^{1}	3.3	3.2×10^{-2}
^{232}Th	9.6×10^{-2}	1.3	3.7×10^{-3}	2.9×10^{-3}
^{233}U	5.8×10^{-1}	7.9	1.7	1.7×10^{-2}
^{234}U	5.8×10^{-1}	7.8	1.7	1.7×10^{-2}
^{235}U	5.2×10^{-1}	7.1	1.5	1.6×10^{-2}
^{236}U	5.4×10^{-1}	7.4	1.6	1.6×10^{-2}
^{238}U	5.1×10^{-1}	7.0	1.5	1.5×10^{-2}
^{238}Pu	2.8×10^{-2}	2.0×10^{-1}	5.7×10^{-2}	3.2×10^{-3}
^{239}Pu	3.1×10^{-2}	2.2×10^{-1}	6.3×10^{-2}	5.6×10^{-3}
^{241}Pu	6.2×10^{-4}	4.6×10^{-3}	1.2×10^{-3}	3.0×10^{-5}
^{241}Am	1.0	7.6	2.2	1.2×10^{-1}
^{242}Pu	3.0×10^{-2}	2.1×10^{-1}	6.0×10^{-2}	3.4×10^{-3}
^{243}Am	1.0	7.6	2.2	1.3×10^{-1}
^{244}Cm	5.6×10^{-1}	3.9	1.2	6.4×10^{-2}

Source: Dunning, D. E., Jr., G. G. Killough, S. R. Bernard, J. C. Pleasant, and P. J. Walsh. 1981. Estimates of Internal Dose Equivalent to 22 Target Organs for Radionuclides Occurring in Routine Release from Nuclear Fuel-Cycle Facilities. ORNL/NUREG/TM-190/V3, Oak Ridge National Laboratory.

Table 6.2. Dose conversion factors for inhalation doses (rem/μCi)

Radionuclide	Organ			
	Total body	Bone	Kidney	Lungs
^{3}H	1.2×10^{-4}	5.6×10^{-5}	1.3×10^{-4}	1.2×10^{-4}
^{14}C	1.4×10^{-5}	8.5×10^{-7}	7.9×10^{-6}	6.2×10^{-6}
^{60}Co	8.2×10^{-2}	5.1×10^{-2}	5.8×10^{-2}	1.3
^{63}Ni	5.7×10^{-3}	1.8×10^{-1}	5.7×10^{-2}	2.3×10^{-2}
^{90}Sr	1.5×10^{-1}	3.2×10^{-1}	3.6×10^{-3}	8.5
^{93}Zr	2.6×10^{-3}	9.4×10^{-2}	2.4×10^{-2}	2.2×10^{-1}
^{99}Tc	8.9×10^{-4}	2.4×10^{-4}	3.1×10^{-4}	5.2×10^{-2}
^{121m}Sn	4.2×10^{-3}	1.2×10^{-1}	2.8×10^{-3}	2.9×10^{-1}
^{123}Te	2.4×10^{-3}	2.3×10^{-3}	1.0×10^{-2}	1.2×10^{-1}
^{137}Cs	3.3×10^{-2}	4.5×10^{-2}	5.1×10^{-2}	1.6×10^{-2}
^{151}Sm	7.1×10^{-3}	1.8×10^{-1}	3.3×10^{-2}	4.6×10^{-2}
^{151}Eu	9.7×10^{-2}	4.8×10^{-1}	5.1×10^{-1}	3.5×10^{-1}
^{226}Ra	4.7	4.9×10^{1}	6.6×10^{-1}	5.6×10^{1}
^{232}U	5.3×10^{1}	3	5.8	8.8×10^{2}
^{232}Th	3.8×10^{1}	1.1×10^{2}	1.1	4.5×10^{2}
^{233}U	1.7×10^{1}	8.0	1.7	5.4×10^{2}
^{234}U	1.6×10^{1}	7.9	1.7	5.4×10^{2}
^{235}U	1.5×10^{1}	7.2	1.5	4.8×10^{2}
^{236}U	1.6×10^{1}	7.5	1.6	5.1×10^{2}
^{238}U	1.5×10^{1}	7.1	1.5	4.8×10^{2}
^{238}Pu	6.0×10^{1}	3.1×10^{2}	9.0×10^{1}	6.1×10^{2}
^{239}Pu	6.7×10^{1}	3.6×10^{2}	1.0×10^{2}	5.8×10^{2}
^{241}Pu	1.2	8.1	2.2	1.1
^{241}Am	6.9×10^{1}	3.7×10^{2}	1.1×10^{2}	6.1×10^{2}
^{242}Pu	6.3×10^{1}	3.4×10^{2}	9.8×10^{1}	5.5×10^{2}
^{243}Am	6.9×10^{1}	3.7×10^{2}	1.1×10^{2}	5.9×10^{2}
^{244}Cm	3.9×10^{1}	1.7×10^{2}	5.2×10^{1}	6.1×10^{2}

Solubility class Y, 1-μm particle size. Source: Dunning, D. E., Jr., G. G. Killough, S. R. Bernard, J. C. Pleasant, and P. J. Walsh. 1981. Estimates of Internal Dose Equivalent to 22 Target Organs for Radionuclides Occurring in Routine Release from Nuclear Fuel-Cycle Facilities. ORNL/NUREG/TM-190/V3, Oak Ridge National Laboratory.

Table 6.3. External dose conversion factors for exposure to contaminated ground surface (millirem/year per $\mu Ci/cm^2$)

Radionuclide	Organ			
	Total body	Bone	Kidney	Lungs
^{60}Co	2.1×10^{6}	2.1×10^{6}	2.0×10^{6}	2.0×10^{6}
^{99}Tc	5.5×10^{2}	8.6×10^{-1}	4.5×10^{-1}	4.9×10^{-1}
^{121m}Sn	5.7×10^{3}	3.7×10^{3}	3.6×10^{3}	2.2×10^{3}
^{123}Te	6.7×10^{3}	5.0×10^{3}	5.0×10^{3}	2.9×10^{3}
^{137}Cs	5.7×10^{5}	6.1×10^{5}	5.3×10^{5}	5.4×10^{5}
^{151}Sm	4.9	2.3	1.5	1.4
^{152}Eu	1.0×10^{6}	1.1×10^{6}	9.7×10^{5}	9.7×10^{5}
^{226}Ra	6.8×10^{3}	9.2×10^{3}	5.8×10^{3}	6.2×10^{3}
^{232}U	9.1×10^{2}	4.8×10^{2}	2.0×10^{2}	2.8×10^{2}
^{232}Th	5.7×10^{2}	3.4×10^{2}	1.5×10^{2}	2.0×10^{2}
^{233}U	4.4×10^{2}	3.7×10^{2}	1.8×10^{2}	2.3×10^{2}
^{234}U	7.1×10^{2}	2.9×10^{2}	1.0×10^{2}	1.7×10^{2}
^{235}U	1.5×10^{5}	2.1×10^{5}	1.3×10^{5}	1.4×10^{5}
^{236}U	6.4×10^{2}	2.4×10^{2}	6.8×10^{1}	1.4×10^{2}
^{238}U	5.7×10^{2}	2.1×10^{2}	5.8×10^{1}	1.2×10^{2}
^{238}Pu	7.7×10^{2}	2.0×10^{2}	2.7×10^{1}	1.2×10^{2}
^{239}Pu	3.4×10^{2}	1.5×10^{2}	4.8×10^{1}	8.9×10^{2}
^{241}Am	2.7×10^{4}	3.7×10^{4}	2.0×10^{4}	2.0×10^{4}
^{242}Pu	6.1×10^{2}	1.7×10^{2}	2.5×10^{1}	1.0×10^{2}
^{243}Am	5.8×10^{4}	8.8×10^{4}	4.6×10^{4}	4.9×10^{4}
^{244}Cm	7.5×10^{2}	2.1×10^{2}	2.1×10^{1}	1.2×10^{2}

Source: Kocher, D. C. 1981. Dose-Rate Conversion Factors for External Exposure to Photons and Electrons. ORNL/NUREG-79, Oak Ridge National Laboratory.

food and all of the drinking water for an individual was obtained at the location of contamination.

In estimating the dose from the inhalation of resuspended contaminated soil, it is assumed that the individual lives on the contaminated land (Sect. 6.2.2). Resuspension factors used for living on the contaminated land (normal activity) are 1×10^{-9} m^{-1} (AEC 1974) and those for mechanically disturbing the land (plowing) are 1×10^{-7} m^{-1} (Healey 1977).

The methodology for determining direct gamma exposure to an individual residing in a house built directly in the waste pits is based on information given in NUREG-0456 (Adams and Rogers 1978). The gamma flux through the concrete floors and walls is approximated by:

$$\phi g = \frac{0.2R}{2N} E_2(b_1) ,$$

where

ϕg = gamma flux (photons/$cm^2 \cdot s$),

R = radionuclide concentration (pCi/cm^3),

N = linear attenuation coefficient for waste soil (cm^{-1}),

$b_1 = \mu_c h$,

μ_c = linear attentuation coefficient for concrete (cm^{-1}),

h = thickness of walls and floors (17 cm), and

E_2 = exponential integral.

The basic equation for the dose rate from a flux of gamma rays is:

$$D = 0.0576 \, \phi_g E_g \left(\frac{\mu}{\rho}^a\right) T_x ,$$

where

D = dose rate (millirem),

ϕg = gamma flux (gammas/cm^2),

E_g = average gamma ray energy (MeV),

$\frac{\mu^a}{\rho}$ = mass absorption coefficient for tissue, and

T_x = exposure time (h).

6.2 DOSE COMMITMENTS

The groundwater medium for trench disposal and the surface water medium for tumulus disposal are generally considered to be the most significant radionuclide migration pathways since they provide a means for exposure of individuals outside the site boundary. These pathways are also of concern for individuals who may unknowingly occupy the site after institutional control and be exposed to radioactivity associated with the waste. To estimate a range of probable impacts, scenarios for pathways that have a potential for restricting the use of the area have been analyzed. The events that might lead to exposure of a land reclaimer are considered in detail.

6.2.1 Nearest Public Drinking Water Source

The Clinch River is the nearest potential public drinking water source that could receive radioactivity from the CWDF. For waste disposal in shallow trenches (the groundwater pathway), leachate that enters the shallow aquifers beneath the site will flow into the alluvium of Ish Creek, New Zion Creek, and Grassy Creek, which discharge into the Clinch River. Potentially, contaminated groundwater in the aquifer could be transported to the Clinch River. Estimates of maximum radionuclide concentrations in the aquifer due to failures of the disposal trenches following site closure and institutional control are given in Table 4.1. These concentrations will be reduced by a factor of 2.4×10^3 as the aquifer flow mixes with the Clinch River flow (Sect. 4.1).

For waste disposal in tumuli (the surface water pathway; see Sect. 2.2.2), leachate could flow overland to on-site creeks that eventually discharge into the Clinch River (Sect. 2.3.3). The dilution factor for leachate that follows this pathway is estimated to be 5.9×10^4 upon complete mixing of water from the creeks with the water of the Clinch River. The resulting concentrations of radionuclides due to the various disposal unit failure events are given in Tables 5.3, 5.4, and 5.5.

6.2.1.1 Early failure event

In this scenario, it is assumed that failure of the design features of the disposal units occurs during the institutional control period (immediately after site closure) and that 10% of the total waste activity migrates out of the disposal units prior to implementation of remedial actions. The resulting maximum radionuclide concentrations in the Clinch River and the corresponding 50-year dose commitments from drinking this water are shown in Table 6.4.

For trench disposal (the groundwater pathway), the total body dose of 0.033 millirem/year is mostly due to ^{3}H (76%) while the highest organ dose of 0.12 millirem/year is to the bone and is 88% attributable to uranium nuclides. These doses are well below the 25 millirem/year whole body and 25 millirem/year organ dose limits (40 CFR 190) used for this study.

The 50-year dose commitments (Table 6.4) that result from tumulus disposal (surface water pathway) are also well below the limits specified in 40 CFR 190 but are considerably larger (about an order of magnitude) than those for trench disposal; the major dose contributors are also different. These differences are due to the retention characteristics of the soil that provide a buffer for some of the radioactivity through sorption of some radionuclides. The total body dose of 2.2 millirem/year results largely from ^{137}Cs (34%); the highest organ dose of 15 millirem/year is to the bone and is largely due to ^{90}Sr (47%).

It can be concluded from these results that, for the waste concentrations considered (Sect. 3), maximum radiation dose commitments to persons outside the site boundary during institutional control will be well below regulatory limits (40 CFR 190).

6.2.1.2 Postinstitutional failure event

During the postinstitutional period, the site would not be maintained. Failure of the design features is assumed to occur rapidly in all disposal units and to result in release of the total waste activity to the groundwater (Sect. 4) and surface water systems (Sect. 5). Table 6.5 shows the maximum resulting radionuclide concentrations in the Clinch River and the corresponding 50-year dose commitments from drinking the river water.

Table 6.4. Maximum 50-year dose commitments from drinking water from a public drinking water supply (Clinch River) after an early failure event[a]

Radionuclide	Max concentration in Clinch River (pCi/L)	Dose (millirem/year)			
		Total body	Bone	Kidney	Lungs
		Trench disposal (groundwater pathway)			
^{3}H	4.1×10^{2}	2.5×10^{-2}	1.4×10^{-2}	3.3×10^{-2}	3.0×10^{-2}
^{14}C	4.2×10^{-2}	5.9×10^{-5}	4.5×10^{-5}	4.1×10^{-5}	3.3×10^{-5}
^{99}Tc	1.1	1.7×10^{-4}	3.6×10^{-4}	4.6×10^{-4}	3.2×10^{-4}
^{234}U	2.7×10^{-3}	1.4×10^{-3}	1.9×10^{-2}	4.2×10^{-3}	4.2×10^{-5}
^{235}U	6.7×10^{-3}	3.2×10^{-3}	4.3×10^{-2}	9.0×10^{-3}	1.0×10^{-4}
^{238}U	6.7×10^{-3}	3.2×10^{-3}	4.3×10^{-2}	9.0×10^{-3}	9.2×10^{-5}
Others		$<1.0 \times 10^{-5}$	$<1.0 \times 10^{-5}$	$<1.0 \times 10^{-5}$	$<1.0 \times 10^{-4}$
Total		3.3×10^{-2}	1.2×10^{-1}	5.6×10^{-2}	3.0×10^{-2}
		Tumulus disposal (surface water pathway)			
^{3}H	5.1×10^{1}	3.1×10^{-3}	1.7×10^{-3}	4.1×10^{-3}	3.4×10^{-3}
^{14}C	4.8×10^{-1}	6.7×10^{-4}	5.1×10^{-4}	4.7×10^{-4}	3.8×10^{-4}
^{60}Co	4.8×10^{-1}	1.5×10^{-3}	1.3×10^{-3}	2.0×10^{-3}	3.0×10^{-3}
^{90}Sr	8.1	5.5×10^{-1}	7.1	3.5×10^{-2}	3.5×10^{-2}
^{93}Zr	2.8×10^{-6}	2.0×10^{-12}	7.6×10^{-11}	1.9×10^{-11}	1.9×10^{-11}
^{99}Tc	9.5×10^{-1}	1.5×10^{-4}	2.5×10^{-4}	3.2×10^{-4}	2.2×10^{-4}
^{121m}Sn	2.2×10^{-4}	1.2×10^{-7}	4.8×10^{-6}	8.0×10^{-8}	1.2×10^{-6}
^{134}Cs	4.8×10^{-1}	2.4×10^{-2}	2.8×10^{-2}	3.5×10^{-2}	5.6×10^{-2}
^{137}Cs	2.0×10^{1}	7.4×10^{-1}	1.0	1.2	1.5
^{151}Sm	4.8×10^{-1}	9.8×10^{-7}	2.4×10^{-5}	4.6×10^{-6}	3.5×10^{-6}
^{192}Ir	9.5×10^{-1}	1.1×10^{-3}	2.9×10^{-3}	2.7×10^{-3}	4.4×10^{-5}
^{234}U	2.4×10^{-3}	1.2×10^{-3}	1.7×10^{-2}	3.7×10^{-3}	3.7×10^{-3}
^{235}U	9.6×10^{-4}	4.6×10^{-4}	6.1×10^{-3}	1.3×10^{-3}	1.4×10^{-5}
^{238}U	5.9×10^{-3}	2.8×10^{-3}	3.8×10^{-2}	7.9×10^{-3}	8.1×10^{-5}
^{238}Pu	7.3×10^{-3}	1.5×10^{-4}	1.1×10^{-3}	3.0×10^{-4}	1.7×10^{-5}
^{239}Pu	2.5×10^{-5}	5.7×10^{-7}	4.0×10^{-6}	1.1×10^{-6}	1.0×10^{-7}
^{241}Am	1.2	8.7×10^{-1}	6.6	1.9	1.1×10^{-1}
^{244}Cm	1.2×10^{-2}	4.9×10^{-3}	3.4×10^{-2}	1.0×10^{-2}	5.6×10^{-4}
Total		2.2	1.5×10^{1}	3.2	1.7

[a]Assumes that design failure occurs at time of site closure. No credit is taken into account for decay of radionuclides prior to site closure.

Table 6.5. Maximum 50-year dose commitment from drinking water from a public drinking water supply (Clinch River) after a postinstitutional failure event

Radionuclide	Max concentration in Clinch River (pCi/L)	Dose (millirem/year)			
		Total body	Bone	Kidney	Lungs
Trench disposal (groundwater pathway)					
^{3}H	1.5×10^{1}	9.1×10^{-4}	5.1×10^{-4}	1.2×10^{-3}	1.1×10^{-3}
^{14}C	4.5×10^{-1}	6.3×10^{-4}	4.8×10^{-4}	4.4×10^{-4}	3.5×10^{-4}
^{99}Tc	1.1×10^{1}	1.7×10^{-3}	3.6×10^{-3}	4.6×10^{-3}	3.2×10^{-3}
^{234}U	2.7×10^{-2}	1.4×10^{-2}	1.9×10^{-1}	4.2×10^{-2}	4.2×10^{-4}
^{235}U	6.6×10^{-2}	3.1×10^{-2}	4.3×10^{-1}	9.0×10^{-2}	1.0×10^{-3}
^{238}U	6.6×10^{-2}	3.1×10^{-2}	4.2×10^{-1}	9.0×10^{-2}	9.2×10^{-4}
Others		$<1.0 \times 10^{-5}$	$<1.0 \times 10^{-5}$	$<2.0 \times 10^{-5}$	$<1.0 \times 10^{-3}$
Total		7.9×10^{-2}	1.05	2.3×10^{-1}	8.0×10^{-3}
Tumulus disposal (surface water pathway)					
^{3}H	1.8	1.1×10^{-4}	6.1×10^{-5}	1.4×10^{-4}	1.3×10^{-4}
^{14}C	4.8	6.7×10^{-3}	5.1×10^{-3}	4.7×10^{-3}	3.7×10^{-3}
^{60}Co	9.9×10^{-6}	3.2×10^{-8}	2.7×10^{-8}	4.2×10^{-8}	6.3×10^{-8}
^{90}Sr	7.2	4.9×10^{-1}	6.3	3.1×10^{-2}	3.1×10^{-2}
^{93}Zr	2.8×10^{-5}	2.0×10^{-11}	7.6×10^{-10}	1.9×10^{-10}	1.9×10^{-10}
^{99}Tc	9.5	1.5×10^{-3}	2.5×10^{-3}	3.2×10^{-3}	2.2×10^{-3}
^{121m}Sn	2.2×10^{-3}	1.2×10^{-6}	4.8×10^{-5}	8.0×10^{-7}	1.2×10^{-5}
^{137}Cs	2.0×10^{1}	7.4×10^{-1}	1.0	1.2	1.5
^{234}U	2.4×10^{-2}	1.2×10^{-2}	1.7×10^{-1}	3.7×10^{-2}	3.7×10^{-2}
^{235}U	9.6×10^{-3}	4.6×10^{-3}	6.1×10^{-2}	1.3×10^{-2}	1.4×10^{-4}
^{238}U	5.9×10^{-2}	2.8×10^{-2}	3.8×10^{-1}	7.9×10^{-2}	8.1×10^{-4}
^{238}Pu	3.3×10^{-2}	6.8×10^{-4}	5.0×10^{-3}	1.4×10^{-3}	7.7×10^{-5}
^{239}Pu	2.5×10^{-4}	5.7×10^{-6}	4.0×10^{-5}	1.1×10^{-5}	1.0×10^{-6}
^{241}Am	1.0×10^{-1}	7.3×10^{-2}	5.5×10^{-1}	1.6×10^{-1}	9.1×10^{-3}
^{244}Cm	2.5×10^{-2}	1.0×10^{-2}	7.1×10^{-2}	2.1×10^{-2}	1.2×10^{-3}
Total		1.4	8.5	1.6	1.6

For the groundwater pathway, the total-body dose of 0.079 millirem/year and the highest organ dose of 1.05 millirem/year (bone) are mostly due to uranium nuclides and are well below the 25-millirem/year total body and organ dose limits (40 CFR 190). Tritium, a major dose contributor for the early failure event (Sect. 6.2.1.1), is not as significant in the postinstitutional period because of its 12.3-year half-life decay.

The results for the surface water pathway show trends similar to those for the early failure scenario; the dose commitments are larger than those for the groundwater pathway but well within regulatory limits (40 CFR 190). The total body dose is 1.4 millirem/year, with ^{137}Cs being the major contributor (53%). The highest organ dose of 8.5 millirem/year is to the bone, with ^{90}Sr being the largest contributor (74%).

Thus for the postinstitutional failure event, individuals outside the site boundary would be exposed to relatively small amounts of radioactivity if the Clinch River were used for a drinking water source. It is noteworthy that the scenario of complete design failure (no containment of the buried radioactivity) is ultraconservative. The likelihood of all trenches degrading and leaching at the same time is a remote possibility. If only a portion of the waste trenches were to degrade and leach simultaneously, the levels of radioactivity in the Clinch River would decrease proportionally.

6.2.2 Direct Intrusion

The direct intrusion event assumes that the intruder builds a house over a disposal unit, lives in the house, eats vegetables grown on the plot and drinks water either from an on-site well (for trench disposal) or from Ish Creek (for tumulus disposal). For this scenario, it is assumed that the basement of the house is 2.4 m deep, 9 m wide, and 15 m long. Assuming that a portion of the 2.0-m trench cap has eroded away over a long period of time, the basement extends 0.9 m into the burial trench. Approximately 122 m^3 (0.9 x 9 x 15 m) of contaminated soils is excavated. The contaminated soil is deposited uniformly over the 61- by 61-m building lot

and mixed with the soil to a depth of 15 cm. The radionuclides in the disturbed layer of soil (0.9 m thick) are mixed uniformly with the existing 15 cm of surface soil (8.37×10^8 g). The estimated concentration of radionuclides in the topsoil is shown in Table 6.6.

The pathways considered for the direct intrusion event are (1) inhalation of suspended particles of radioactive dust, (2) ingestion of vegetables grown on the plot, (3) exposure to direct gamma radiation, and (4) ingestion of contaminated water.

6.2.2.1 Inhalation of suspended particles of contaminated dust

It is assumed that the intruder lives on the contaminated land and spends 80 h/year mechanically disturbing the soil (digging or plowing) and, for the remainder of the year, is subjected to suspended dust particles by wind and normal activity. Table 6.6 provides the concentrations of radionuclides in the soil. A suspension factor of 1×10^{-7} m^{-1} is assumed during mechanical disturbance of the soil and 1×10^{-9} m^{-1} during normal activity. The doses to the inadvertent intruder from the inhalation of contaminated dust are shown in Table 6.7. The maximum total-body dose is 5.3 millirem, while the highest organ dose of 130 millirem is to the lungs, with about 80% attributable to uranium radionuclides.

6.2.2.2 Ingestion of vegetables produced on contaminated soil

It is assumed that the intruder has a vegetable garden in the contaminated top soil (Table 6.6) and that the concentration of radionuclides in the soil remains unchanged; i.e., the radionuclides do not migrate beyond the root zone (15 cm) of the plants. For the maximally exposed individual, it is assumed that 10% of all the food consumed is produced from this garden (NRC 1976). The doses from this pathway are shown in Table 6.8. Approximately 16% of the total-body dose of 210 millirem is due to uranium and 52% to ^{14}C. The highest organ dose of 920 millirem is due primarily to uranium radionuclides (48%) and ^{90}Sr (36%).

Table 6.6. Concentration of radionuclides for the direct intrusion event at 100 years following closure

Radionuclide	Concentration in burial trench		Concentration in topsoil	
	pCi/cm^3	pCi/g	pCi/cm^3	pCi/g
^{3}H	2.3×10^{2}	1.5×10^{2}	5.0×10^{1}	3.3×10^{1}
^{14}C	5.5×10^{2}	3.7×10^{2}	1.2×10^{2}	8.0×10^{2}
^{60}Co	1.0×10^{-3}	6.7×10^{-4}	2.2×10^{-4}	1.5×10^{-4}
^{63}Ni	1.3×10^{-4}	8.7×10^{-5}	2.8×10^{-5}	1.9×10^{-5}
^{90}Sr	8.8×10^{2}	5.8×10^{2}	1.9×10^{2}	1.3×10^{2}
^{93}Zr	3.9×10^{2}	2.6×10^{2}	8.5×10^{1}	5.6×10^{1}
^{99}Tc	2.0×10^{4}	1.3×10^{4}	4.3×10^{3}	2.8×10^{3}
^{121m}Sn	7.4×10^{1}	4.9×10^{1}	1.6×10^{1}	1.1×10^{1}
^{123}Te	9.1×10^{-2}	6.6×10^{-2}	2.0×10^{-2}	1.4×10^{-2}
^{137}Cs	2.4×10^{3}	1.6×10^{3}	5.2×10^{2}	3.5×10^{2}
^{151}Sm	2.1×10^{2}	1.4×10^{2}	4.6×10^{1}	3.0×10^{1}
^{152}Eu	1.6×10^{-3}	1.1×10^{-3}	3.5×10^{-4}	2.4×10^{-4}
^{226}Ra	7.7×10^{-1}	5.1×10^{-1}	1.7×10^{-1}	1.1×10^{-1}
^{232}U	4.9×10^{-5}	3.3×10^{-5}	1.1×10^{-5}	7.2×10^{-6}
^{232}Th	1.9×10^{2}	1.3×10^{2}	4.1×10^{1}	2.8×10^{1}
^{233}U	5.3	3.5	1.2	7.6×10
^{234}U	2.4×10^{1}	1.6×10^{1}	5.2	3.5
^{235}U	4.0×10^{1}	2.7×10^{1}	8.8	5.9
^{236}U	1.7×10^{2}	1.1×10^{2}	3.7×10^{1}	2.4×10^{1}
^{238}U	1.1×10^{3}	7.5×10^{2}	2.5×10^{2}	1.7×10^{2}
^{238}Pu	5.4	3.6	1.2	7.8×10^{-1}
^{239}Pu	3.9	2.6	8.5×10^{-1}	5.7×10^{-1}
^{241}Pu	1.0×10^{-3}	6.7×10^{-4}	2.2×10^{-4}	1.5×10^{-4}
^{241}Am	1.6×10^{1}	1.1×10^{1}	3.5	2.4
^{242}Pu	1.3	8.7×10^{-1}	2.8×10^{-1}	1.9×10^{-1}
^{243}Am	9.4×10^{-3}	6.3×10^{-3}	2.0×10^{-3}	1.4×10^{-3}
^{244}Cm	4.1×10^{-1}	2.9×10^{-7}	8.9×10^{-2}	5.9×10^{-2}
^{249}Cf	4.3×10^{-7}	2.9×10^{-7}	9.3×10^{-8}	6.3×10^{-8}

Table 6.7. Fifty-year dose commitment from inhalation[a] of suspended soil radioactivity

Radionuclide	Dose (millirem)			
	Total body	Bone	Kidney	Lungs
	Normal activity			
^{232}Th	5.7×10^{-1}	1.7	1.5×10^{-2}	6.8
^{235}U	4.9×10^{-2}	2.3×10^{-2}	4.9×10^{-3}	1.6
^{236}U	2.2×10^{-1}	1.1×10^{-1}	2.2×10^{-2}	7.1
^{238}U	1.4	6.5×10^{-1}	1.4×10^{-1}	4.5×10^{1}
Other	5.0×10^{-1}	8.0×10^{-1}	3.0×10^{-1}	7.0
	Plowing or digging			
^{232}Th	5.2×10^{-1}	1.5	1.5×10^{-2}	6.2
^{235}U	4.6×10^{-2}	2.2×10^{-2}	4.3×10^{-3}	1.4
^{236}U	2.0×10^{-1}	1.0×10^{-1}	2.0×10^{-2}	6.5
^{238}U	1.3	5.8×10^{-1}	1.3×10^{-1}	4.0×10^{1}
Other	4.6×10^{-1}	7.3×10^{-1}	2.7×10^{-1}	6.4
Total	5.3	6.2	9.2×10^{-1}	1.3×10^{2}

[a]Assumed breathing rate of 8000 m^3/year.

Table 6.8. Fifty-year dose commitment from eating vegetables grown in contaminated soil[a]

Radionuclide	Dose (millirem)			
	Total body	Bone	Kidney	Lungs
^{14}C	1.1×10^{2}	6.8×10^{1}	6.2×10^{1}	4.8×10^{1}
^{90}Sr	2.6×10^{1}	3.3×10^{2}	1.6	1.6
^{99}Tc	1.9×10^{1}	3.2×10^{1}	4.1×10^{1}	2.9×10^{1}
^{137}Cs	2.2×10^{1}	3.0×10^{1}	3.4×10^{1}	4.5×10^{1}
^{232}Th	1.5	2.0×10^{1}	5.0×10^{-1}	5.0×10^{-1}
^{233}U	1.4×10^{-1}	1.9	4.1×10^{-1}	4.1×10^{-2}
^{234}U	6.6×10^{-1}	8.9	1.9	2.0×10^{2}
^{235}U	1.0	1.4×10^{1}	2.9	3.1×10^{-2}
^{236}U	4.1	5.7×10^{1}	1.2×10^{1}	1.2×10^{-1}
^{238}U	2.7×10^{1}	3.6×10^{2}	7.9×10^{1}	7.9×10^{-1}
Other	1.0	2.0	1.0	5.0×10^{-1}
Total	2.1×10^{2}	9.2×10^{2}	2.3×10^{2}	1.3×10^{2}

[a]Assumed that 10% (28 kg/year) of the vegetables are home grown.

6.2.2.3 Direct gamma exposure

Two scenarios for direct gamma exposure to the maximally exposed individual were examined. The first assumes that the house is built on the contaminated top soil and the individual spends 90% of the time indoors and 10% outdoors. Table 6.9 shows the external gamma doses. The maximum total-body dose is 160 millirem/year and is due almost entirely to ^{137}Cs (97%).

In the second scenario, the basement of the house is sunk directly into one of the burial trenches. The basement walls and the floor, built of 17-cm-thick concrete, are in direct contact with the waste concentrations of the burial pit. The maximally exposed individual spends 50% of the time in the basement. The dose calculations are based on the methodology described in NUREG-0456 (Adams and Rogers 1978). The total-body dose, shown in Table 6.10, of 140 millirem/year is essentially all due to ^{137}Cs.

6.2.2.4 Drinking water

For the groundwater pathway (trench disposal), it is conservatively assumed that the well is drilled near the burial pit, where the highest concentration of radionuclides in the aquifer would occur (see Table 4.1). The 50-year dose commitments from drinking the well water are shown in Table 6.11. The total-body 50-year dose commitment of 190 millirem is almost entirely due to the uranium nuclides. The bone dose is much higher (2500 millirem/year), with the uranium nuclides also being the major dose contributors.

By comparison, the dose commitments from the surface water pathway (tumulus disposal) are roughly an order of magnitude larger than those for the groundwater pathway (Table 6.11). The total-body dose is 3000 millirem/year; ^{90}Sr (37%) and ^{137}Cs (53%) are the major contributors. Similarly, ^{90}Sr (78%) and ^{137}Cs (12%) are major contributors to the bone dose of 18,000 millirem/year.

It should be kept in mind that the peak concentrations used to calculate these doses decrease as time increases. For example, for the

Table 6.9. Maximum dose to the individual living in a house built on contaminated soil[a]

Radionuclide	Dose (millirem)			
	Total body	Bone	Kidney	Lungs
^{137}Cs	1.6×10^{2}	1.8×10^{2}	1.5×10^{2}	1.5×10^{2}
^{235}U	7.4×10^{-1}	8.3×10^{-1}	6.7×10^{-1}	6.8×10^{-1}
Other	<5.0	<5.0	<5.0	<5.0
Total	1.6×10^{2}	1.8×10^{2}	1.5×10^{2}	1.5×10^{2}

[a]It is assumed that the individual spends 10% of the time out of doors and 90% indoors. The house affords a factor of 2 shielding from contaminated surface soil.

Table 6.10. Maximum dose to the individual in the basement of a house built in the waste trench[a]

Radionuclide	Dose (millirem)			
	Total body	Bone	Kidney	Lungs
^{137}Cs	1.4×10^{2}	1.5×10^{2}	1.3×10^{2}	1.3×10^{2}
^{235}U	1.1×10^{-1}	1.3×10^{-1}	1.1×10^{-1}	1.0×10^{-1}
Other	<1.0	<1.0	<1.0	<1.0
Total	1.4×10^{2}	1.5×10^{2}	1.3×10^{2}	1.3×10^{2}

[a]It is assumed that the individual spends 50% of the time in the basement.

Table 6.11. Maximum 50-year dose commitment to the inadvertent intruder through the drinking water pathways

Radionuclide	Max concentration in drinking water[a] (pCi/L)	Dose (millirem/year)			
		Total body	Bone	Kidney	Lungs
Trench disposal (groundwater pathway)					
^{3}H	3.6×10^{4}	2.2	1.2	2.8	2.7
^{14}C	1.0×10^{3}	1.4	1.1	1.0	7.8×10^{-1}
^{99}Tc	2.5×10^{4}	3.9	8.2	10.4	7.2
^{234}U	6.5×10^{1}	3.4×10^{1}	4.6×10^{2}	1.0×10^{2}	1.0
^{235}U	1.6×10^{2}	7.5×10^{1}	1.0×10^{3}	2.1×10^{2}	2.3
^{238}U	1.6×10^{2}	7.3×10^{1}	1.0×10^{3}	2.1×10^{2}	2.2
Others		$<2.0 \times 10^{-2}$	$<2.0 \times 10^{-2}$	$<5.0 \times 10^{-2}$	<.3
Total		1.9×10^{2}	2.5×10^{3}	5.3×10^{2}	1.7×10^{1}
Tumulus disposal (surface water pathway)					
^{3}H	3.9×10^{3}	2.4×10^{-1}	1.3×10^{-1}	3.0×10^{-1}	2.9×10^{-1}
^{14}C	1.0×10^{4}	1.4×10^{1}	1.1×10^{1}	1.0×10^{1}	7.8
^{60}Co	2.2×10^{-2}	7.1×10^{-5}	6.0×10^{-5}	9.3×10^{-5}	1.4×10^{-4}
^{90}Sr	1.6×10^{4}	1.1×10^{3}	1.4×10^{4}	6.8×10^{1}	6.8×10^{1}
^{93}Zr	6.0×10^{-2}	4.3×10^{-8}	1.6×10^{-6}	4.1×10^{-7}	4.1×10^{-7}
^{99}Tc	2.1×10^{4}	3.3	5.4	7.0	4.8
^{121m}Sn	4.8	2.6×10^{-3}	1.1×10^{-1}	1.7×10^{-3}	2.6×10^{-2}
^{137}Cs	4.3×10^{4}	1.6×10^{3}	2.1×10^{-3}	2.6×10^{-3}	3.2×10^{-2}
^{234}U	5.3×10^{1}	2.6×10^{1}	3.7×10^{2}	8.1×10^{1}	8.1×10^{1}
^{235}U	2.1×10^{1}	1.0×10^{1}	1.3×10^{2}	2.8×10^{2}	3.1×10^{-1}
^{238}U	1.3×10^{2}	6.1×10^{1}	8.3×10^{2}	1.7×10^{2}	1.8
^{238}Pu	7.2×10^{1}	1.5	1.1×10^{1}	3.1	1.7×10^{-1}
^{239}Pu	5.5×10^{-1}	1.2×10^{-2}	8.8×10^{-2}	2.4×10^{-2}	2.2×10^{-3}
^{241}Am	2.4×10^{2}	1.8×10^{2}	1.3×10^{3}	3.9×10^{2}	2.2×10^{1}
^{244}Cm	5.6	2.2	1.6×10^{1}	4.7	2.7×10^{-1}
Total		3.0×10^{3}	1.8×10^{4}	7.6×10^{2}	3.4×10^{3}

[a]Taken from Table 4.1 (postinstitutional failure scenario) for groundwater pathway and from Table 5.5 (Ish Creek Station 2) for surface water pathway.

groundwater pathway, the concentrations of the uranium nuclides will decrease by a factor of about 8 at roughly 200 years after the occurrence of the peak (250 years), thus effecting a similar reduction in the dose commitments. Because of the exponential decay assumed for leachate migration from the tumuli (Figure 5.3), the peak concentrations decrease much more rapidly, reaching a value of 12.5% of the peak in less than 50 years. Hence, an intruder would be exposed to the maximum doses for only a finite period of time.

6.2.2.5 Cumulative dose to the inadvertent intruder

Potential radiation doses to an inadvertent intruder are summarized in Table 6.12. For the trench disposal option (use of on-site wells for drinking water), the cumulative total-body dose is 560 millirem/year, with ingestion of vegetables (37.5%), direct exposure (28.5%), and drinking water (33.9%) accounting for more than 99%. The highest cumulative organ dose (3600 millirem/year) is to the bone; more than 69% results from the drinking water pathway.

The cumulative radiation doses for the tumulus disposal option (use of Ish Creek for drinking water) is much higher (an order of magnitude). The total-body and bone (highest organ) doses are 3,400 and 19,000 millirem/year, respectively (Table 6.12). Ingestion of water from Ish Creek accounts for more than 90% of the dose.

For each disposal option, the cumulative radiation doses exceed the limits included in DOE Order 5480.1A (500 millirem/year to the whole body and 1500 millirem/year to the bone), but they are less than the limit of 5000 millirem/year (whole body) for workers at a nuclear facility. However, it should be noted that, for trench disposal, an inadvertent intruder would incur cumulative doses somewhat lower than those given here since the maximum doses from individual pathways would not be incurred simultaneously. The estimated concentration of radioactivity in groundwater (Table 4.1), for example, reaches a maximum about 50 years after the trench failure event, which is assumed to occur immediately after the institutional control period. Because of the decay of relatively short-lived radionuclides (^{137}Cs, for example) during the buildup of

Table 6.12. Summary of maximum doses to the intruder living in the disposal area after lifting of institutional controls[a]

Pathway	Dose (millirem)			
	Total body	Bone	Kidney	Lungs
Ingestion of vegetables[a]	2.1×10^2	9.2×10^2	2.3×10^2	1.3×10^2
Inhalation of resuspended particles[b]	5.3	6.2	9.2×10^{-1}	1.3×10^2
External dose from contaminated soil[c]	1.6×10^2	1.8×10^2	1.5×10^2	1.5×10^2
Drinking Water				
Onsite well	1.9×10^2	2.5×10^3	5.4×10^2	1.7×10^1
(Ish Creek)	(3.0×10^3)	(1.8×10^4)	(7.6×10^2)	(3.4×10^3)
Total	5.6×10^2	3.6×10^3	9.2×10^2	4.3×10^2
	(3.4×10^3)[d]	(1.9×10^4)	(1.1×10^3)	(3.8×10^3)

[a]Assumed that 10% of vegetables consumed are grown in contaminated soil.

[b]Based on a resuspension rate of 10^{-9} m^{-1} for normal activity and 10^{-7} m^{-1} for mechanically disturbing the soil.

[c]Assumed that individual spends 10% of the time out of doors and 90% indoors; house gives factor of 2 shielding.

[d]Total for surface water pathway (tumulus disposal). Ish Creek is the assumed drinking water source.

radioactivity in the groundwater system, the dose from the other intrusion pathways would be less. Moreover, radioactivity (leachate) that enters the groundwater would not contribute to the other pathways, thus further reducing the cumulative dose. This phenomena is too complex to model for the assumed exposure pathways.

6.3 RADIOLOGICAL IMPACT ON BIOTA OTHER THAN HUMANS

No guidelines concerning acceptable limits of radiation exposure have been established for protection of species other than humans. It is generally agreed, however, that, like those for humans, radiation exposure limits are also conservative for other species (Auerbach 1971, Blaylock and Witherspoon 1976, Frigerio et al. 1975, and Garner 1971). Doses to terrestrial biota such as birds and mammals from surface and airborne radionuclides would be quite similar to those calculated for humans and would arise from the same dispersion pathways and considerations.

As long as the earthern cover (including the cap) is intact (NRC 1982), there is little likelihood that burrowing animals would reach the contaminated soil in the burial trenches. For example, the woodchuck (Marmota monax), one of the deeper burrowing animals, burrows to a depth of about 1.5 m. Thus, as long as the earth cover and cap are intact, this pathway of exposure to external gamma radiation is not significant.

After institutional controls are lifted, should erosion reduce the cover to less than 1.5 m, then the total-body dose to burrowing animals from external gamma radiation, conservatively assuming that they spend all of their time underground in the burial waste pit, would be approximately 1 rem/year for the soil concentration included in Table 6.6.

It is assumed that measures taken to maintain safe radiological protection limits for humans would also preclude adverse radiological impacts to resident animals.

7. DISCUSSION OF RESULTS

This section discusses the results of the foregoing analysis in view of the conservative assumptions made and the likelihood of occurrence of the exposure pathways. The capacity of the CWDF on the West Chestnut Ridge Site for disposal of low-level radioactive waste anticipated from the three Oak Ridge facilities is also estimated.

7.1 INTERPRETATION OF RESULTS

Disposal of low-level radioactive waste at the West Chestnut Ridge Site could result in radiation doses to persons outside the DOE Oak Ridge Reservation and to inadvertent intruders onto contaminated areas of the site following site closure and institutional controls.

If the disposal units consist of shallow trenches, as currently planned, leachate from the waste would be expected to contaminate portions of the shallow aquifers that underlie the site. Contaminated groundwater could surface at a nearby wet-weather drainage path (downstream of New Zion Creek) and in the streambeds of Ish Creek and, to a lesser extent, Grassy Creek. Contaminated water could subsequently be transported to the Clinch River, which is a potential future source of drinking water for the general public. This pathway is considered to be the principal means of exposure of persons outside the reservation.

On the basis of the results of this study and site characterization data (Ketelle and Huff 1984), the predicted zone of groundwater contamination in the shallow aquifers is within an area shown in Fig. 7.1. Independent of waste burial concentrations, the area is defined by Ish Creek to the east, Tennessee Highway 95 and Bear Creek Road to the north, Grassy Creek to the northwest, the Clinch River and the approximate western limit of the subterranean portion of New Zion Creek to the west, and the Clinch River to the south. There appears to be adequate buffer space within this area and the disposal site so that human use of the region beyond the buffer zone could be unrestricted.

For aboveground disposal using the tumulus concept, the surface water pathway is considered to be the principal means of exposure of persons

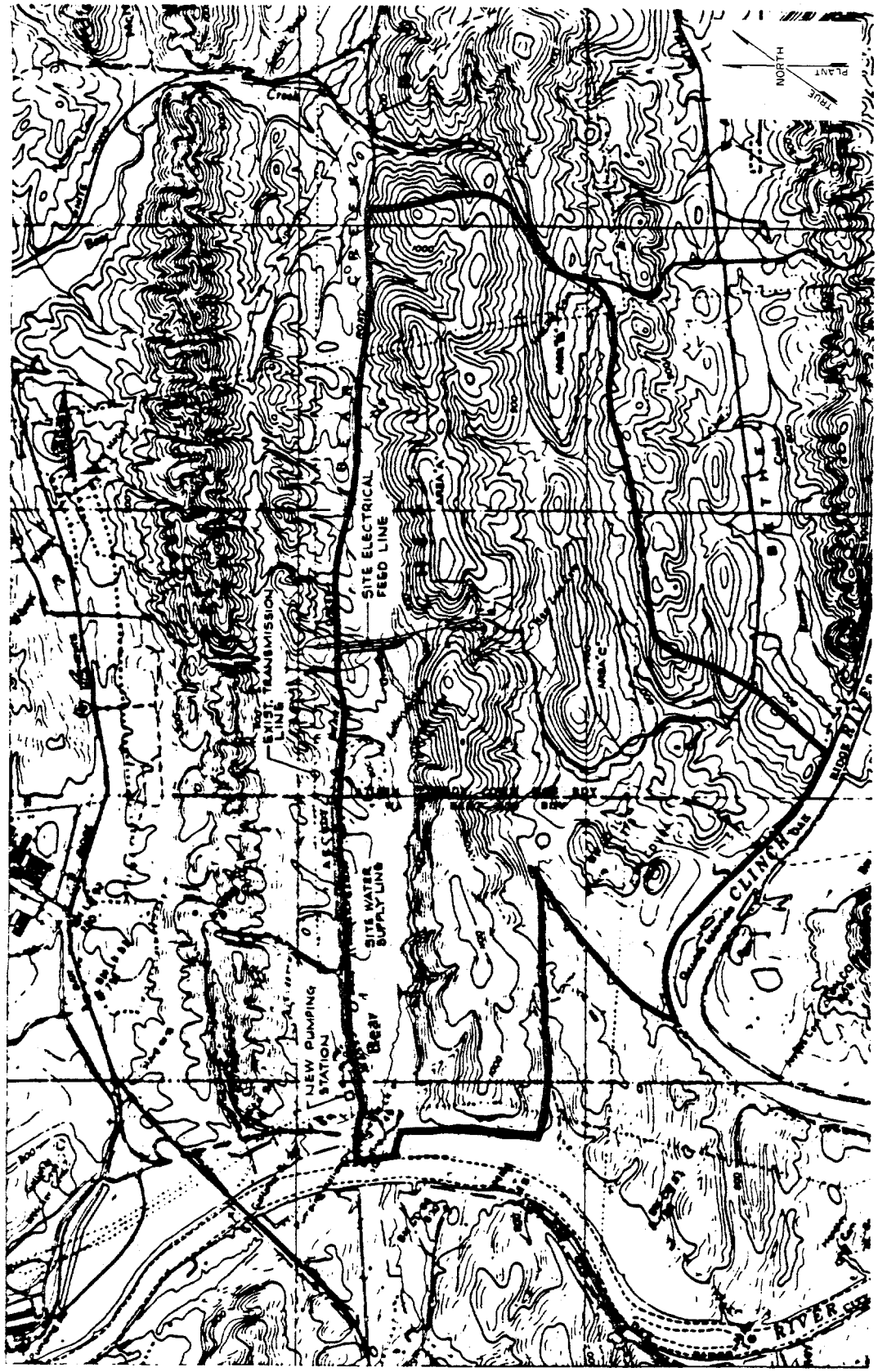

Fig. 7.1. Recommended buffer zone based on predicted area of potential groundwater contamination.

outside the reservation. Ish Creek and the wet-water drainage paths downstream of New Zion Creek could be contaminated and also convey radioactivity to the Clinch River. Groundwater contamination could also occur but at levels considerably less than those considered in the groundwater pathways analysis. As a result of the potential for groundwater contamination, the buffer zone shown in Fig. 7.1 would also be appropriate for aboveground disposal.

The maximum radiation doses to persons who might use the Clinch River as a source of drinking water are estimated to be less than 4% of the regulatory limits for trench disposal and less than 40% for tumulus disposal. These results are based on the conservative assumption of complete loss of containment of each disposal unit simultaneously, a remote possibility as noted below. Thus, there is reasonable assurance that persons outside the Oak Ridge Reservation would not be exposed to hazardous levels of radioactivity.

For an inadvertent intruder, the maximum radiation doses are estimated to exceed regulatory limits (especially for the bone). However, trench disposal, having attendant doses an order of magnitude less than those for tumulus disposal, showed greater protection for the intruder. The source of drinking water (on-site well for trench disposal and Ish Creek for tumulus disposal) was found to be the most important contributor to the dose. In reaching conclusions concerning risk to an inadvertent intruder on the basis of these results, however, consideration must be given to uncertainties and conservatisms associated with the analysis.

First, the initial leachate concentrations in the disposal units were calculated on the assumption that all of the water-diversion engineering features fail at the same time, resulting in complete saturation of the waste and a relatively high rate of leachate generation over a short period. Such a wetting event and leaching characteristics produce the maximum source term for the water pathways analysis. While it seems highly improbable that all of the waste would be saturated and would leach in this manner, no defensible arguments currently exist for assuming that this could not occur. Even for these conservative assumptions, which are equivalent to an extreme failure event, the radiation doses do not exceed the limits for workers at nuclear facilities except for the bone dose for

the tumulus disposal technology. It is also noteworthy that the 500-millirem/year whole-body and 1500-millirem/year organ limits for an inadvertent intruder would be exceeded for only a finite length of time. For trench disposal, the time period is estimated to be on the order of tens of years well after the assumed failure event (Table 4.1). For tumulus disposal, the period of exposure would be reduced to a few years. Thus, an inadvertent intruder would have to occupy the site during this critical period and ingest contaminated water at the site. Since the events of (1) simultaneous and complete failure of all the disposal units and (2) the residence of an intruder at the site for a limited number of years are extraordinarily unlikely in combination, a more realistic estimate of the intruder doses could be those for the early failure scenario.

Second, the validity of the exposure pathways considered for an inadvertent intruder (especially on the Oak Ridge Reservation) is questionable. It is assumed that the area will remain rural in character and that an individual might build a house on the disposal unit, plant a garden, and drill a well into a contaminated portion of the aquifer for water or consume contaminated surface water. Although this generic scenario was used by the NRC to establish maximum disposal concentrations for low-level radioactive waste (NRC 1982), these events appear to be remote for the reservation. Even if all records concerning the waste disposal site were destroyed and an inadvertent intruder were to establish a place of residence and a garden on a disposal unit, it is improbable that drinking water would be taken from an on-site well or from Ish Creek. The Clinch River would be a more dependable source.

Finally, except for the water pathway, the unit waste mass assumed for the direct intrusion pathways produces conservative source terms. The analysis of early failure or postinstitutional control failure does not give credit for any decay that occurs during the operation of the site. For long-lived nuclides this is not important, but for short-lived nuclides such as ^{3}H, ^{60}Co, ^{90}Sr, ^{121m}Sn, ^{137}Cs, and ^{192}Ir, this could amount to one or more half-lives of decay. The analysis of the postinstitutional control period assumes that no reduction in the radionuclide inventory occurs during institutional control. Any reduction in the inventory of the

waste by migration during institutional control would correspondingly reduce the concentrations of radionuclides in the intruder pathways during the postinstitutional control period. Furthermore, the typical unit waste mass for the direct intruder analysis was based on the maximum expected local concentrations in the waste streams. Consequently, the total quantity of some radionuclides was increased significantly. Obviously, these assumptions markedly increase the probability that an inadvertent intruder would contact significant levels of radioactivity. Even so, the maximum doses are not prohibitively high compared to regulatory limits. After a decay period of a hundred years or so, major dose contributors such as ^{137}Cs and ^{90}Sr are not of concern.

Thus, in view of these factors -- the containment and isolation afforded persons outside the site boundary and the relatively small health risk (DOE 1983b) associated with the estimated doses to an inadvertent intruder -- there is reasonable assurance that the site can accommodate the subject waste streams (Appendix B) without significant environmental impact. On the other hand, for trench disposal (the proposed method), the pathways analysis relies on the assumption that the site is properly operated during the waste disposal period and is properly controlled and maintained for at least 100 years following closure of the disposal units. Major emphasis was placed in the conceptual design (Ebasco 1984) on the construction, maintainance, and operation of the trench engineering features--side wall drains, drainage blankets, trench caps, and surface water diversion systems. A part of the conservatism built into the analysis relies on the proper operation and maintainance of these systems throughout the institutional control period. The degree of conservatism involved in drawing conclusions from the results of this analysis would be reduced by any significant modification of the proposed plans for disposal, design, or maintainance of the site that would increase the probability of occurrence of the scenarios used in the analysis.

7.2 SITE CAPACITY

The proposed design for the CWDF consists of shallow trenches Sect. 2.2.1). Of interest are the maximum concentrations of radionuclides

that can be accepted for disposal in the trenches. Preliminary waste acceptance criteria developed for the CWDF are included in Appendix C. The concentrations included in the criteria are those given in 10 CFR 61 for Class A waste since the facility is designed for such waste. The results of this pathways analysis provide a basis for adjusting these concentrations for the West Chestnut Ridge Site prior to finalizing the waste acceptance criteria.

The source terms used for the pathways analysis are based on the typical unit waste masses given in Tables 3.2 and 3.3. The concentrations of radionuclides in the waste are less than those for Class A waste in 10 CFR 61 and, in some instances, the radionuclides are not major contributors to the radiation doses for the limiting pathways. Hence, it is possible to increase the concentration of these radionuclides to the Class A limit without effecting any noticeable change to the attendant radiation doses. The site capacity was determined on this basis.

For each radionuclide expected to be disposed of at the CWDF, the maximum permissible concentration for shallow land burial was chosen to correspond to the smaller of the concentrations for Class A waste or the concentration that results in limiting radiation doses for selected pathways. These concentrations are listed in Table 7.1. The exposure pathways for an inadvertent intruder were used to determine the maximum concentrations since the dose commitments to an inadvertent intruder could approach regulatory limits. The pathway to the general public was not considered limiting for determining site capacity since the potential dose commitments would be a small fraction of those allowed under present regulations. The maximum concentrations of each radionuclide listed in Table 7.1 were calculated using the maximum permissible dose of 5000 millirem/year, which is equivalent to the occupational dose limit. This limit was used because only a few individuals could be considered to be capable of receiving doses from the intruder scenario considered in the pathways analysis, and the direct intrusion scenario is a worst-case event.

The potential doses that could be received from short-lived nuclides are expected to vary considerably with time. The doses calculated in the pathways analysis, which assumes a 100-year institutional control period, provide reference values for comparison of the potential impact of these nuclides. All of the short-lived radionuclides of concern (^{3}H, ^{90}Sr,

Table 7.1. Maximum permissible concentrations of radionuclides in CWDF waste for shallow land burial

Radionuclide	Max permissible concentration (Ci/m^3 unless noted)
Short-lived radionuclides	
Total of all nuclides with half-lives less than 5 years	700
^{3}H	40
^{60}Co	700
^{63}Ni	3.5
^{90}Sr[a]	0.04
^{137}Cs[a]	0.6
^{244}Cm	100 nCi/g
Long-lived radionuclides	
^{14}C[a]	0.025
^{99}Tc	0.05
Uranium	0.0007
Alpha-emitting transuranic nuclides with half-lives greater than 20 years	100 nCi/g
^{241}Pu	350 nCi/g
^{242}Cm	2000 nCi/g

[a]Limit for individual packages. All other limits are averaged over the disposal site.

and ^{137}Cs) are to be disposed of with high concentrations. In the very unlikely event that institutional control ceases before the end of the 100-year period assumed for the reference values used in this study, the potential impact due to these nuclides could be higher than predicted. Consequently, in addition to good waste disposal and waste packaging practices, long-term maintainance and control of the site are prime factors for keeping the impact of these nuclides as low as reasonably achievable within the prescribed regulatory standards.

The potential doses from the major long-lived radionuclides (^{14}C, ^{99}Tc, ^{234}U, ^{235}U, and ^{238}U) are not expected to vary with time, and the length of the institutional control period is of little significance for the impact of these nuclides. Improvements in waste packaging and waste isolation (e.g., fixation in a concrete matrix) to reduce both the leach rate of these nuclides from the unstabilized waste and the probability of exposure through direct intrusion into the waste could result in higher disposal concentrations without greater impact than predicted for the values used in this study. For some of the long-lived radionuclides, Table 7.1 gives estimates of the higher disposal concentrations for stabilized and isolated waste (assuming an order of magnitude decrease in the leach rate).

The waste projected for disposal at the CWDF originates from a variety of processes. Individual waste packages or masses may contain very different assortments of radionuclides. Arrangement of the packages in the trenches may vary considerably from one trench to another and from one location to another in a single trench. On the basis of the results of this analysis, the concentrations specified in Table 7.1 for ^{90}Sr, ^{137}Cs, and ^{14}C are recommended as maximum permissible waste concentrations in individual packages. The values given for the other radionuclides are recommended as maximum permissible waste concentrations averaged over the entire site. After waste emplacement, as assumed for the pathways analysis, the trenches should be backfilled with an approximately equal volume of soil, and the mix of waste material and soil should be compacted to a 50% nominal void fraction and maintained to minimize the effects of subsidence.

The maximum concentrations (Table 7.1) are by no means exact, but they are reasonable approximations for the conservatisms built into the analyses. Higher or lower concentrations may be considered, depending on the level of significance attached to the source terms and especially the intruder-type scenarios used in the analyses. In the course of performing the pathways analysis, field and laboratory data have been relied upon and applied conservatively. Conservatism has been built into the analysis when assumptions concerning future events had to be made. Some of this conservatism may prove to be unnecessary when additional data become available. An area that largely determines the conclusions of the pathways analysis is the characterization of the waste. The quantity, nature, form, and radioactivity content of the waste provide the basis for determining the source terms. For less conservative source terms (e.g., lower leachate generation rates), higher burial concentrations may be used with a prediction of no greater impact on human health and safety than that for for the values used in this study. Investigations of the long-term solubility and leaching of radionuclides from both the unstablized and stabilized wastes could provide lower leachate concentrations and quantities than those used in this analysis. Reductions in the concentration and quantity of the leachate available for transport would enable the maximum concentrations suggested for the waste acceptance criteria to be increased above those identified in Table 7.1. Similarly, if less weight is given to the event where an inadvertent intruder has a home and garden on a trench and uses contaminated water for drinking water, higher waste burial concentrations could be used but with correspondingly greater off-site impact. The suggested radionuclide concentrations appear to provide the proper balance of protection for the general public and inadvertent intruders.

8. CONCLUSIONS

This study has considered the radiological implications of disposal of low-level radioactive waste anticipated from the Oak Ridge National Laboratory, the Y-12 Plant, and the Oak Ridge Gaseous Diffusion Plant at the proposed CWDF on the West Chestnut Ridge Site. The capacity of the site to safely accommodate the waste has been determined. In the absence of specific information, assumptions have been made and parameters have been selected to produce conservative results. On the basis of the analyses and findings of this study, the following major conclusions have been reached.

1. The West Chestnut Ridge Site is suitable for the disposal of low-level radioactive waste. Site monitoring and maintenance would allow for the potential of radioactive contamination to be as low as reasonably achievable and less than regulatory standards.

2. Both the shallow land burial (trench) and aboveground (tumulus) disposal methods can be deployed effectively at the CWDF to contain and isolate the subject waste. The proposed shallow land burial disposal method, however, will provide more effective containment because of the sorptive nature of the soil for some of the radionuclides.

3. Persons outside the Oak Ridge Reservation could be exposed to small quantities of radioactivity if the Clinch River were used for drinking water purposes. Conservative estimates of maximum radiation doses to these persons suggest that the doses would be well below regulatory limits.

4. Depending on the nature and time of a direct intrusion event, persons who inadvertently occupy the site could be exposed to radioactive materials that result in doses that approach the regulatory limits. However, the likelihood of any such intrusion would be remote and, in any case, would involve only a limited number of individuals.

5. Natural features of the West Chestnut Ridge Site and the proposed design coupled with state-of-the-art operational and maintenance procedures are sufficient to safely accommodate the waste streams (Appendix B) anticipated from the three Oak Ridge plants. Independent of waste burial concentrations, the zone of groundwater contamination is defined by Ish Creek to the east, Tennessee Highway 95 and Bear Creek Road to the north, Grassy Creek to the northwest, the Clinch River and the approximate western limit of the subterranean portion of New Zion Creek to the west, and the Clinch River to the south.

Appendix A

SOLUBILITY CALCULATION RESULTS FOR RADIONUCLIDES EXPECTED TO BE DISPOSED OF AT THE CENTRAL WASTE DISPOSAL FACILITY*

This appendix documents the results of solubility calculations for elements representing selected radionuclides that are expected to be disposed of at the Central Waste Disposal Facility (CWDF). The calculated limits represent upper bounds for concentrations that could occur in the groundwater at the CWDF site and may be used for assessment of possible environmental impacts from site development and use.

A.1 BACKGROUND

In principle, the calculation of solubility limits is based on the thermodynamics of chemical solutions and their reactions with solid phases. It is a fundamental premise of chemical thermodynamics that a system will spontaneously react so as to approach the minimum total Gibbs free energy. Solubility calculations are intended to estimate the solution composition and the solid phases that would be present at the condition of lowest possible Gibbs free energy, or equilibrium. A chemical system that is not at equilibrium will spontaneously react to approach equilibrium.

These principles may be applied to the CWDF under either of two sets of conditions or assumptions: (1) chemical equilibrium may be assumed or (2) reactions between solutions and solid phases that contain the radionuclide may be described. In the first instance, it must be assumed that the waste/ leachate/groundwater chemical system will be at equilibrium and that the description used for that system in making calculations has been accurate. In fact, although it is likely that many of the possible reactions in the system will be reasonably near equilibrium, it is also

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*Written by N. E. Cutshall of ORNL's Environmental Sciences Division (Cutshall to Pin Memo, May 22, 1984) as part of the CWDF geochemical program.

well established that there are some reactions that proceed only very slowly under the low temperatures of the CWDF. For example, few silica or silicate minerals are likely to be at equilibrium. Reactions between the solid oxide forms of manganese and the soluble reduced species rarely approach equilibrium. The presence of minute surficial coatings on solid phases may prevent equilibrium between the solids and the solutions. These coatings may be iron/manganese/aluminum oxides or organic matter or a mixture thereof. Because of these complications, it is probably unjustified to assert that equilibrium calculations absolutely predict the geochemistry of the site. Rather, such procedures describe the geochemical endpoint toward which the system can be expected to react.

If, on the other hand, the waste radionuclides are in a known solid chemical form, then it is reasonable to assert that calculations of the solubility limits for that compound are valid. For example, solid radium sulfate will dissolve into groundwater only up to the solubility limits. Thus, limiting solubility calculations may be applied with much higher confidence where the solid phases that contain the radionuclide are known. Solubility calculations need to account for reactions among dissolved chemical species. Under conditions where the solutes interact to form complexes or ion pairs, the total concentration of an element or radionuclide in a saturated solution can be considerably increased. The geochemical codes used are designed to account for the effects of solution reactions on total solubility limits. Reactions among the constituents known to be present or specified as part of the input data are included in the computation process.

Through most of this discussion, the term "mobility" is used interchangeably with "solubility". In cases where there is a significant transport of particulate material, such as in turbid streams, there may also be transport of precipitated or adsorbed radionuclides.

Actual field experience in radionuclide or trace element transport by natural waters indicates that mobility is generally overestimated by limiting solubility calculations, provided that the solution and waste chemical factors described above are properly taken into account. Solubility calculations do not include the formation of solid solutions or the adsorption of the trace components by immobile solid phases. In real-world systems, these latter processes typically provide effective mobility

controls that are orders of magnitude lower than those calculated from solubility considerations. Incorporation of sorption limits to mobility is considerably more difficult than computation of solubility limits. Sorption will vary greatly according to substrate composition and condition. For example, the penetration of both anions and cations through the Tarklin soils that characterize the sinkhole zones of Chestnut Ridge, is considerably greater than penetration of the same species through the Fullerton soils that are common elsewhere along the ridge. This observation is frequently explained by the greater leaching and depletion of the reactive iron and manganese oxide coatings from the Tarklin soils. The significance to the CWDF site is that sorption assessments for soils that occur along principal groundwater flow pathways cannot rely entirely upon sorption data for soil samples taken from locations away from these highly leached zones.

A.2 COMPUTATIONAL PROCEDURES

The calculations for all nuclides were made using the PHREEQE code (Parkhurst et al. 1980). Uranium calculations were also made using the MINTEQ code because the authors of that code have been especially thorough in validating the uranium data base. MINTEQ is a new code developed at Battelle Pacific Northwest Laboratory (Felmy et al. 1982).

Two sets of solution composition data for the CWDF groundwater were defined on the basis of chemical analysis of CWDF groundwater (Seeley and Kelmers 1984) and are shown in Table A.1. The composition of Sample 2 was used in all computations. Because Sample 2 contains higher concentrations of ligands that might increase solubility through complexation, it is expected that computations with Sample 1 would yield the same or lower solubility limits. All computations were run for 25°C and an assumed pH of 7.0 (Eh = +414 mV). The assumed Eh is representative of values that might be measured in waters that are in contact with the atmosphere. Organic decomposition could cause the trench solutions and soil to become anaerobic and reducing. In the absence of direct analytical data, however, the assumption of an aerated condition this near the surface seems most reasonable for the bulk soils. Each of the codes used includes an internal

Table A.1. Input groundwater chemical data

Parameter	Value or concentration (mg/L)	
	Sample 1	Sample 2
pH	5.7	5.7
Sodium	2.8	15.0
Calcium	0.5	2.1
Magnesium	0.17	1.1
Manganese	0.14	1.5
Barium	0.03	0.09
Silicon (SiO_2)	6.6	6.8
Chlorine	2.0	24.0
Carbon(CO_2)	6.5	8.7

thermodynamic data base. For the uranium computations using MINTEQ, the standard MINTEQ data base was used. In the case of computations using PHREEQE, the standard PHREEQE data base was supplemented either by data that had been compiled by Early et al. (1982) or by calculations from data in Garrels and Christ (1965). In the case of radium, technetium, thorium, uranium, neptunium, and plutonium, all reactions from Early et al. were used. For zirconium, tin, samarium, europium, and americium, only reactions with H_2O, OH^-, H^+ and the carbonate species were included.

To estimate solubility limits, the codes simulate the precipitation or dissolution of a selected solid and determine the solution species that would exist at equilibrium and at saturation with respect to that solid. It is necessary to assume some initial concentration of the elements involved in the precipitation reaction (the default concentration is zero). The simulated reaction affects the concentrations of each of the reactants and products in the dissolution reaction. For example, the radium sulfate dissolution adds not only radium but also sulfate to the solution. Thus, although the initial solution composition data do not include sulfate, to estimate the solubility limit for radium sulfate the codes must add data for this constituent. Otherwise, there would not be a mathematical solution to the equilibrium equations. Perhaps more significantly, in the case of several of the oxides or hydroxides, the simulated dissolution of the solids affects pH. Consequently, the computed solubility limits are sometimes for solutions that are considerably more basic than CWDF water. If the system pH is buffered by soil minerals, then the pH effects of the simulated dissolution may be ameliorated. The net result is that several metals would be considerably more soluble in a solution buffered at a pH of 5.7 than in a solution at the higher pH computed in the simulation. Furthermore, the degree of the effect on pH depends in part on the assumed initial concentration of the element of interest. Choosing a very high concentration of a metal that precipitates as a hydroxide would lead to lowering the pH, whereas choosing a very low concentration of the same element would raise the pH through dissolution. Where the simulated reaction significantly affects pH, that fact is noted in Table A.2. The final pH for the particular simulation is also indicated. For the elements noted, the solubility limit would be different at a different pH, possibly markedly different.

Table A.2. Computational results

Element	Solid phase assumed	Dominant species	Solubility Limit (mol/L)	Comments[a]
Hydrogen				See discussion.
Beryllium	BeO	Be^{++}	6×10^{-10}	
Carbon	$CaCO_3$	H_2CO_3		See discussion.
Phosphorus	$Ca_3(PO_4)_3$			See discussion.
Manganese	MnO_2	Mn^{+2}	1×10^{-4}	Buffers pH. (8.3)
Iron	Fe_2O_3	Fe^{+2}	1×10^{-13}	
Cobalt				See discussion.
Krypton				See discussion.
Strontium	$SrCO_3$	Sr^{+2}		See discussion.
Niobium				See discussion.
Zirconium	$ZrSiO_4$	$Zr(OH)_4$	7×10^{-12}	
Technetium	TcO_2	TcO_4-2	6×10^{-4}	Buffers pH. (3.3)
Ruthenium				See discussion.
Iodine				See discussion.
Cesium				See discussion.
Tin	SnO_2	$SnO(OH)^+$	2×10^{-14}	
Cerium				See discussion.
Promethium				See discussion.
Samarium	$Sm(OH)_3$	Sm^{+3}	5×10^{-5}	Buffers pH. (7.0)
Europium	$Eu(OH)_3$	Eu^{+3}	1×10^{-4}	Buffers pH. (6.9)
Iridium				See discussion.
Polonium				See discussion.
Lead	$PbCO_3$	Pb^{++}	6×10^{-5}	
Radium	$RaSO_4$	Ra^{+2}	7×10^{-6}	
Thorium	ThO_2	$Th(OH)_4$	8×10^{-15}	
Uranium	Schoepite	$(UO_2)_3(OH)_5^+$	4.5×10^{-5}	
Neptunium	NpO_2	NpO_2^+	3×10^{-7}	
Plutonium	PuO_2	PuO_2^{+2}	1×10^{-12}	
Americium	AmO_2OH	$Am(OH)_4$	4×10^{-8}	
Curium				See discussion.
Californium				See discussion.

[a]Numbers in parentheses indicate the final pH values for the simulation.

The nuclides for which calculations were desired are listed in Table A.2. Relevant solubility data for some elements are not available; therefore, solubility calculations for those elements are of dubious meaning. The results shown in Table A.2 indicate the estimated solubility limits and the species that is expected to predominate in solution.

A.3 DISCUSSION OF RESULTS BY ELEMENT

The elemental results show that solubility constraints will keep some nuclides at very low concentrations in the CWDF groundwater. Other nuclides may not be effectively immobilized by simple insoluble phases, but they may be effectively prevented from escaping by sorption. While the computations described above do not include sorption as a retarding mechanism, the following discussion will indicate the nuclides for which waste management experience shows that "soluble" nuclides are effectively immobilized by sorption.

Hydrogen: All isotopes of hydrogen will behave essentially identically; for example, tritium migration in aqueous form will not be retarded significantly more than as nontritiated waste. Tritium is extremely mobile. The only constraints for tritium will be physical containment in the waste form.

Beryllium: In soil or freshwater systems, beryllium is relatively immobile. While BeO is not a naturally occurring phase, sorption of Be^{++} ions on naturally occurring oxides is expected to be at least equally effective in preventing movement.

Carbon: Carbon is ubiquitous in soils as carbonates and complex mineral compounds. Radiocarbon exchange in such forms is very slow; hence, any migration of radiocarbon can be expected as in the chemical form received. If the radiocarbon is contained in organic forms, however, these may be persistent for extremely long times; if they are soluble, they may be highly mobile. In the absence of specific information on the chemical form of the carbon, it is not possible to infer any solubility limit.

Phosphorus: No computations were done for phosphorus. Note that the half-life of the disposed isotope is only 14 d. Phosphorus added to soil as a fertilizer is known to bind to oxides of iron and may reasonably be expected to be retarded.

Manganese: The mobility of manganese in oxidizing systems is low, owing to the formation of the insoluble oxides. Under reducing conditions, formation of the Mn^{++} could increase mobility, but such conditions are not expected at the CWDF beyond the immediate trench boundaries.

Iron: Under usual soil conditions, iron prevails as Fe_2O_3 and associated complex compounds; such phases are among the least soluble mineral species (see Table A.2).

Cobalt: Cobalt, like the other two elements of its subgroup (manganese and iron), prevails in soils as low-solubility oxides.

Krypton: Krypton, like the other noble gases, is insoluble in water. It will not be chemically retarded by waste components or geologic constituents at the CWDF, and its mobility should be considered to be very high. Restrictions on mobility will come from physical containment.

Strontium: Radioisotopes of strontium by far exceed the hazard index (mobility/radiologic hazard) of any other nuclide. At the low pH of CWDF groundwater, no reasonable solubility limits can be assumed. Strontium radioisotope retardation will result strictly from sorption, probably on reversible ion exchange sites of the clay minerals and iron oxides present. In more alkaline environments, strontium can be contained. Treatment of soils with bases increases their sorption of strontium.

Niobium: No calculations were performed for niobium. Because of its short (35-d) half-life, ^{95}Nb (daughter of ^{95}Zr) should not present a containment problem. The geochemistry of niobium is similar to that of zirconium; the extreme insolubility of the oxides of these elements and their potential for sorption will effectively retard movement.

Zirconium: Zirconium oxide or silicate solubility is exceedingly low.

Technetium: In oxidizing systems, technetium forms the pertechnetate ion, which is relatively mobile in soils. Reduction to Tc(IV) and precipitation as TcO_2 is the solubility control, but its effectiveness in oxidizing systems is poor. Acidic conditions favor reduction. Mobility approaching that of tritium has been observed at other disposal sites.

Ruthenium: No calculations were done for ruthenium. The existence of mobile forms of ruthenium radioisotopes in wastes that have been produced from nitric acid solutions is widely experienced. These complexes are kinetically inert, and they persist in solution far from equilibrium.

Practical experience shows that ruthenium isotopes are relatively mobile in soil/groundwater systems.

Iodine: No calculations were made for iodine. Iodine, as iodide or iodate, will be highly mobile, and no solid phases are expected to be formed. Sorption will be minimal.

Cesium: No calculations were made for cesium. Solubility limitations for cesium were expected to be ineffective, but sorption on illitic clays has previously been shown to quantitatively contain cesium radioisotopes. Mobility at other Oak Ridge sites is immeasurably slow.

Tin: Tin forms highly insoluble oxides and is expected to be kept below even the oxide solubility limit by sorption.

Rare Earths--Cerium, Promethium, Samarium, Europium: Calculations were done only for samarium and europium. The hydroxides of these elements are relatively insoluble, and the rare earths are highly susceptible to sorption. In previous waste disposal experience, no rare earth isotope has been found to be mobile. In comparison with other radionuclides, the rare earths are of negligible concern.

Iridium: Iridium, like the noble metals platinum and osmium, will occur in nonreactive metallic form.

Polonium: ^{210}Po (132-d half-life) is the daughter of ^{210}Pb (22-year half-life), which was not on the list of nuclides of concern. It is sorbed by oxides or iron and by other surfaces. Mobility in natural systems appears to be low. Polonium is similar in chemistry to phosphorus.

Lead: Although lead was not on the list of nuclides of concern, because of the relationship to polonium, it was included in the calculations. Insoluble phases may be either oxides or carbonate.

Radium: Solubility of the radium sulfate is low; and the element is not prone to complex formation. Compared to other heavy elements such as thorium and reduced uranium, radium is relatively mobile.

Thorium: Thorium has only the tetravalent oxidation state in natural systems. Thorium carbonates and ThO_2 are only slightly soluble. The mobility of thorium is among the very lowest of all the elements.

Uranium: Uranium is most mobile in oxidizing, alkaline systems with carbonate ion present. The pH at the CWDF is too low to allow carbonate complexation. Either higher or lower pH would probably increase the solubility of uranium.

Neptunium: Neptunium, like uranium, would be more soluble at higher pH where carbonate complexation would become a factor.

Plutonium: Plutonium has never been found to be mobile in groundwater, and the solubility calculations are consistent with this observation. The calculations were done omitting polymeric species.

Americium: Americium is slightly more mobile than plutonium, but it still forms quite insoluble oxide phases.

Curium: No calculations were done for curium. Waste disposal experience with curium shows that its isotopes are the most mobile transuranics. Complexation may be a factor. There are only limited thermodynamic data for curium; little, if any, chemical modeling has been done.

Californium: No data are known to exist for californium. No calculations were performed.

Appendix B

WASTE STREAM CHARACTERIZATION FOR THE CENTRAL WASTE DISPOSAL FACILITY*

Each of the three plants operated by the Department of Energy on the Oak Ridge Reservation produces low-level radioactive wastes at an approximately constant rate, in proportion to the ongoing research, development, and/or production activity. In addition to the quantity of waste generated at baseline rates, nonroutine activities will produce (1) sludges removed from waste holding ponds at the Y-12 Plant (Y-12) and Oak Ridge Gaseous Diffusion Plant (ORGDP) (processed and fixed), (2) contaminated ash from incineration of polychlorinated biphenyls (PCBs), and (3) equipment and materials for disposal as a result of decontamination and decommissioning (D&D) of surplus facilities.

The following waste stream characterization differentiates the baseline- and campaign-originated wastes for purposes of facility performance analysis. The values permit integrations for estimation of facility capacity, operational lifetime, and for pathways analysis. It is recognized, however, that such integrations may or may not correspond to the approximately 40-year nominal loading period for the facility because baseline rates may change in the future and other not-yet-identified nonroutine requirements may be placed on the Central Waste Disposal Facility (CWDF).

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*This appendix is based on the characterization compiled by R. E. Thoma in a memorandum to L. D. Bates, June 21, 1984, and issued by the CWDF Program on June 30, 1984, for concurrence. Although the data are characteristic of the projected streams, revisions of the quantities, schedules, and specific waste forms may be expected both before and after operation of the facility begins.

B.1. VOLUME PROJECTIONS

B.1.1 Summary Projections

B.1.1.1 Baseline rate projection summary (Values are ft^3/year.)

ORNL* (X-10 site)	59,000	Baled, bulk, etc.
	1,000	Asbestos-contaminated bulk
Y-12	119,070	As compacted; 30% max.
	6,930	Asbestos contaminated bulk
	108,000	West End Treatment Facility sludges
	32,000	CPCF* sludges
	41,000	Mercury-contaminated sludges and ion exchange resins; begins FY-88
ORGDP	19,000	Thickener sludge; begins FY-87
	8,000	Miscellaneous sludges
	4,000	Asbestos-contaminated LLW*
	5,000	Miscellaneous bulk materials
	30,000	TSCA* ash
	7,000	TSCA grout
Total	440,000	

B.1.1.2 Nonroutine rate projection summary (Values are ft^3/year except as noted otherwise)

ORNL (X-10 site)	40,000	From D&D activities; begins in FY-1989
Y-12	370,000	See Sect. B.1.2.2
ORGDP	4,000 tons†	Contaminated scrap metal
Total	410,000	

B.1.1.3 Baseline and nonroutine summary

ORNL (X-10 site)	90,000
Y-12	677,000
ORGDP	73,000
Total	840,000

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*ORNL = Oak Ridge National Laboratory;
CPCF = Central Pollution Control Facility;
LLW = Low-level Waste;
TSCA = Toxic Substances Control Act.

†See Sect. B.1.2.3; volume was not estimated nor included in current total.

B.1.2 Individual Plant Projections

B.1.2.1 ORNL (X-10 site)

Combustible (bulk)	Percentage	ft^3/year
Paper and cloth	18	18,000
Plastics	9	9,000
Rubber	2	2,000
Wood	0.5	500
Other	0.5	500
Total	30	30,000

Noncombustible/noncompactible		
Ferrous metal	24	24,000
Nonferrous metal	8	8,000
Glass and ceramics	16	16,000
Concrete	1.5	1,500
Soil	1	1,000
Resin	1.5	1,500
Other	18	18,000
Total	70	70,000

The two preceding waste categories will take the following forms:

Baled waste, 20,000 ft^3/(compaction ratio = 8) — 2,500 ft^3

Bulk waste, minimum volume — 6,000 ft^3

- o asbestos contaminated: 1,000 ft^3
- o large equipment: 5,000 ft^3

Compactible, combustible — 41,500 ft^3

- o shreddable: 29,500 ft^3
- o from D&D: 12,000 ft^3

Noncompactible — 50,000 ft^3

- o 27,000 less 5,000 (equipment): 22,000 ft^3
- o noncombustible (from D&D): 28,000 ft^3

The D&D waste breakdown into noncompactible/compactible is assumed to be in the same ratio as the overall ORNL waste breakdown into noncombustible/combustible of 70,000/30,000 = 2.33.

Decontamination of surface impoundments at ORNL will result in the production of fixed or dewatered sludges, residues, and contaminated soils that will be disposed of at the CWDF. Volumes and associated radioactivity of the materials are not defined at this time.

B.1.2.2 Y-12

Baseline rates

Solid wastes

- Total volume is approximately 126,000 ft^3/year (see Table B.1 for detailed listing).
- 30-40% may be shreddable.
- This shredded waste will be compacted into bales 60 x 30 x 40 in.) weighing 1500 to 5000 lb. Packaging will meet U.S. Department of Transportation requirements, which might involve only wrapping bales in plastic.
- Approximately 7000 ft^3/year of LLW contaminated with asbestos may be delivered to the CWDF.

Sludges

- Sludges produced from treatment of wastes at the Y-12 West End Treatment Plant: 108,000 ft^3/year.
- Sludges produced from treatment of miscellaneous wastes at the Central Pollution Control Facility at ORGDP: Y-12 origin, 32,000 ft^3/year.
- Mercury-contaminated sludges and ion exchange resins: 41,000 ft^3/year, beginning in FY-88.

Figure B.1 indicates estimates of generation rates of waste sludges from all sources. The increases indicated during FY-85 and -86 pertain to increasing generating capacity of the treatment plants during the onset of their operations.

Table B.1. PROPOSED Y-12 SOLID WASTE DELIVERIES TO THE CWDF

Waste	Isotope	Uncompctd Vol.(cf)	Compctd Vol.(cf)	Total Wgt.(lbs)	Percent Isotope	Wgt. of Isotope (grams)	Curies/yr	Material Composition/ Waste Description
Carbon	D 238				.998	634329	0.32	Scrap Pieces & Dust
Carbon	D 235				.002	1271	0.00	
Carbon	D 234				.000016	10	0.00	
Carbon	Total	8100	5670	140000	1	635600	0.32	
Mixed Chips	D 238				.2495	1814663	0.91	Mixed Non Uranium Machine
Mixed Chips	D 235				.0005	3637	0.00	Turnings Contaminated With
Mixed Chips	D 234				.000004	29	0.00	Depleted Uranium Turnings,
Mixed Chips	Total	80000	56000	1602000	.25	1818300	0.91	Way Oils and Machine Coolants
Glass	D 238			40000	.00998	1812	0.00	Rinsed Chemical Containers
Glass	D 235				.00002	4	0.00	Panes Ground Glass,
Glass	D 234				1.6E-7	.029056	0.00	Flourescent Light Bulbs,
Glass	E 238			40000	.0009	163	0.01	Lab. Glassware, etc.
Glass	E 235				.009	1634	0.11	
Glass	E 234				.0001	18	0.00	
Glass	Total	5400	3780	80000	.01	3632	0.13	
Filters	D 238			0	.0998	54371	0.03	Filter Media Contaminated
Filters	D 235				.0002	109	0.00	With Particulates
Filters	D 234				1.6E-6	.87168	0.00	
Filters	Total	29700	20790	120000	.1	54480	0.03	
Asbestos	D 238			38000	.0998	113273	0.01	Insulation Containing
Asbestos	D 235				.0002	227	0.00	Asbestos, Including Some
Asbestos	D 234				1.6E-6	2	0.00	Metal Piping and Duct
Asbestos	E 238			38000	.009	10215	0.11	
Asbestos	E 235				.089	102150	1.09	
Asbestos	E 234				.001	1135	0.01	
Asbestos	Total	9900	6930	76000	0.1	34504	1.22	
Drums	D 238				.0998	10421	0.01	30 and 55 Gallon Metal Drums
Drums	D 235				.0002	21	0.00	Contaminated With Coolant and
Drums	D 234				1.6E-6	.167072	0.00	Waste Oil Residual
Drums	Total	4100	2870	23000	.1	10442	0.01	
Press Area Debris	D 238				.998	2034383	1.02	Stainless Heat Shield,
Press Area Debris	D 235				.002	4077	0.00	Steel Shot, Furnace Brick,
Press Area Debris	D 234				.000016	33	0.00	Rubber Pads, Blotter
Press Area Debris	Total	11000	7700	449000	1.0	2038460	1.02	Paper, Vacuum Hose, etc.
Roll Mill	D 238				4.99	2718552	1.36	Insulation Brick, Anodes,
Roll Mill	D 235				.01	5448	0.00	Salt Bath Tankage, Floor
Roll Mill	D 234				.00008	44	0.00	Sweepings and Blotter Paper
Roll Mill	Total	4300	3010	120000	5	2724000	1.36	Contaminated with K, Na, and Li Carbonate Salts
Debris	D 238				.0998	135928	0.07	Includes Waste From Maintanance
Debris	D 235				.0002	272	0.00	and Construction Activities:
Debris	D 234				.000016	22	0.00	Insulation, Paint, Cans, Duct,
Debris	Total	27000	18900	300000	.1	136200	0.07	Piping, Equipment, Combustibles, etc.

Table B.1. (continued)

Waste	Isotope	Uncomptd Vol.(cf)	Comptd Vol.(cf)	Total Wgt.(lbs)	Percent Isotope	Wgt. of Isotope (grams)	Curies/yr	Material Composition/ Waste Description
Lab Waste	D 238			300	.00998	14	0.00	Uranium Metal and Oxide Samples,
Lab Waste	D 235				.00002	.02724	0.00	Glass, Carbon, Paper, Ceramics,
Lab Waste	D 234				1.6E-6	.002179	0.00	and Metal Salvage Materials
Lab Waste	N 238			300	.00993	14	0.00	
Lab Waste	N 235				7.2E-5	.098064	0.00	
Lab Waste	N 234				5.8E-7	.00079	0.00	
Lab Waste	Total	150	105	600	.01	27	0.00	
Aerosol Cans	Total Depleted U	150	105	500	0	0	0.00	Pressurized Cans and Small Cylinders with Residual Gases and Aerosols
Magnesium Chips	Total Depleted U	150	105	3200	0	0	0.00	Magnesium Metal Machine Turnings
Carbon Foam	Total Depleted U	50	35	9750	0	0	0.00	Composition: Urethane Foam Resin, Toluene Di-Isocyanate and Alcohol. Waste Including Carbon Foam Pieces, Lard Cans, Rags, Paper Buckets, etc.
Totals		180000	126000	2923050		7455645	5.07	

NOTES: 1. 1983 data used for the quantification of the wastes as a 1st approximation.
2. Isotopic information based on incomplete data which is to be refined during FY 1984, and made available by the end of FY 1984.
3. Volume reduction assumes a maximum of 30 percent reduction for all listed waste.

June 7, 1984

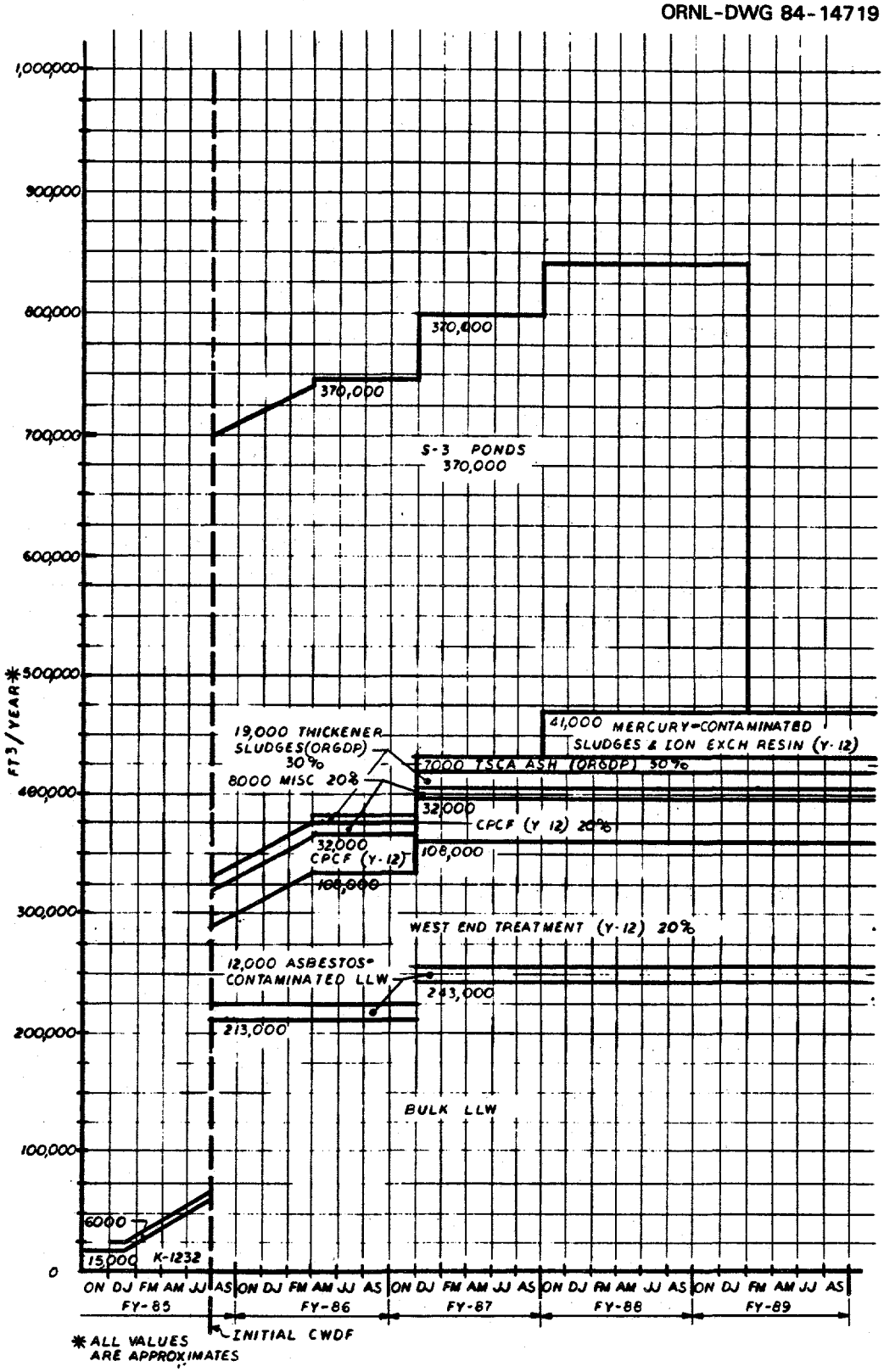

Fig. B.1. Projected volumes and forms of low-level wastes for disposition in the Central Waste Disposal Facility.

The rates of receipt of wastes at the CWDF may be proportional to but are not equal to the generation rates.

Nonroutine waste

Currently, it is projected that 370,000 ft^3/year of pond sludges from the S-3 waste ponds will be fixed in a grout matrix and transported to the CWDF. The waste will have a unit weight of approximately 150 lb/ft^3 and a moisture content of approximately 40%. The grout will contain no free liquid. The volume of material to be received has been calculated from the following assumptions: (1) sludges from all four S-3 ponds will be fixed and disposed of at the CWDF, (2) 5 ft of soil beneath each pond will be fixed and disposed of at the CWDF, and (3) fixation and disposal will occur over a 42-month period. A schedule for disposal of the sludges is shown in Fig. B.1.

B.1.2.3 ORGDP

Baseline rates

- LLW contaminated with asbestos: ~4,000 ft^3/year; contains 3 lb uranium (0.001 Ci/year).
- Fixed and dewatered sludge (thickener sludges): 19,000 ft^3/year; contains 22 lb uranium (0.03 Ci/year) and 18 g ^{99}Tc (0.3 Ci/year).
- Miscellaneous sludges generated from a variety of sources: 8,000 ft^3/year.
- Miscellaneous materials: 5,000 ft^3/year, containing 37.5 lb uranium (0.2 Ci/year) and 170 g ^{99}Tc (2.8 Ci/year).
- Ash and slag from TSCA incinerator: ~35,000 ft^3/year, containing 160 lb uranium (0.06 Ci/year).

Nonroutine waste

- Miscellaneous materials (nonprocessable) from ORGDP Scrap Yard: 3,000 tons/year; contains 600 lb uranium (0.2 Ci/year) and 2 Ci ^{99}Tc. This material will be produced only in 1985.

B.2. ISOTOPIC COMPOSITION, ACTIVITY, AND CHEMICAL FORM

B.2.1 ORNL (X-10 Site)

At a generation rate of 60,000 ft^3/year, of which, ~1,500 ft^3 will have been compacted (compation ratio: ~8); the total annual activity will be 345 Ci (increasing to 500 Ci by 1989), the sources of which are:

Isotope	Ci/year	Half-life	Chemical form
^{14}C	2	5770 years	
^{60}Co	2	5.27 years	
^{134}Cs	2	2.2 years	
^{137}Cs	82	30 years	Chloride
^{3}H	210	12 years	Water
^{192}Ir	4	74 d	Metallic
^{85}Kr	2	10.4 years	Elemental
^{151}Sm	2	93 years	
^{121}Sn	1	23 years	
^{90}Sr	34	28 years	Fluoride carbonate
^{93}Zr	2	9.5×10^5 years	
Total	345		

In addition, ~40,000 ft^3/year of concrete and other building materials from facility modification and decommissioning operations will be generated each year beginning in 1988, with proportionately smaller amounts in intervening years (1984-1987). It is assumed that the baseline activity for each radionuclide listed above will increase in proportion to the increase in waste volume. The total associated activity will be about 500 Ci/year. The maximum yearly contribution transuranics (TRUs) that might be expected from D&D operations on transuranic-contaminated facilities is as follows:

^{238}Pu	30 mCi/year
^{239}Pu	10 mCi/year
^{241}Am	50 mCi/year
^{244}Cm	50 mCi/year
Total	140 mCi/year

In addition to the TRU nuclides listed above, about 100 mCi/year of mixed TRU nuclides (mainly ^{239}Pu) would be expected because of the change in the definition of TRU Waste in DOE Order 5820.

B.2.2 Y-12

B.2.2.1 Baseline rates (See Sect. B.1.2.2)

B.2.2.2 Nonroutine wastes

The Y-12 Plant will transfer sludges from the S-3 and other storage ponds to the CWDF on the schedule shown in Sect. B.1.2.2. Sludges from these ponds will contain the materials and activities shown in Tables B.2 through B.4.

B.2.3 ORGDP

See Sect. B.1.2.3. In addition to these materials, accumulated sludges and contaminated ash from incinerated polychlorinated biphenyls (PCBs) will be transferred to the CWDF during the initial 4 to 6 years of its operation. The cumulative activity from these sources is as listed below (see Sect. B.1.2.3 for chemical forms).

Isotope	Ci/year
^{238}U	
^{236}U	
^{235}U	0.7
^{234}U	
^{239}Pu	femtocuries
^{237}Np	femtocuries
^{99}Tc	2.2
^{232}Th	trace

Table B.2. Analysis of the S-3 disposal pond sediments[a]
(All values are in micrograms per gram (µg/g) of dry solids.)

Test	Pond			
	Southeast	Northeast	Southwest	Northwest
Silver	7.3	17.0	2.1	4.1
Aluminum	41,897.8	41,854.0	59,034.9	21,643.8
Arsenic	14.8	32.5	26.0	21.7
Boron	70.0	98.7	138.6	55.7
Barium	359.4	428.6	285.9	337.3
Beryllium	16.4	1.3	2.9	1.4
Calcium	3,962.9	1,005.5	1,952.2	894.3
Cadmium	<0.6	<0.6	<0.6	<0.6
Cerium	48.0	45.9	73.3	72.3
Cobalt	3.3	1.6	1.4	<0.01
Chromium	163.9	75.9	135.1	48.5
Copper	145.3	136.4	111.1	128.2
Iron	92,031.0	89,500.0	26,284.2	8,232.8
Gallium	11.9	1.6	33.5	30.5
Hafnium	19.4	4.9	14.3	14.0
Mercury	12.0	1.7	0.88	0.21
Potassium	8,000.3	11,070.6	23,762.6	8,307.9
Lanthanum	25.5	37.7	42.1	45.7
Lithium	46.3	46.9	35.1	29.2
Magnesium	2,341.1	2,614.7	4,437.0	1,593.7
Manganese	112.0	108.1	63.1	45.9
Molybdenum	191.7	103.8	113.7	30.1
Sodium	1,429.5	1,768.9	1,993.5	2,041.0
Neodymium	136.8	62.0	30.9	75.6
Nickel	98.8	73.9	60.6	62.9
Phosphorus	6,896.4	1,333.5	2,454.8	2,296.9
Lead	119.7	198.1	155.0	207.0
Scandium	5.6	8.1	8.4	4.6
Selenium	<0.2	<0.2	<0.2	<0.2

Table B.2. (continued)

Test	Pond			
	Southeast	Northeast	Southwest	Northwest
Silicon	40.9	37.5	666.0	22.7
Strontium	40.3	46.0	62.3	66.5
Thorium	271.7	150.0	196.0	529.4
Titanium	3,630.5	5,172.1	5,120.1	5,206.1
Uranium	620.0	280.0	410.0	300.0
Vanadium	63.1	61.5	64.7	26.5
Yttrium	8.4	12.5	12.1	11.6
Zinc	91.2	95.0	56.3	34.0
Zirconium	1,472.0	817.7	1,077.4	3,366.0

[a]Results obtained by inductively coupled plasma (ICP) method.

Table B.3. Radionuclide analyses of the S-3 disposal pond sediment[a]

Radionuclide	Pond			
	Southeast	Northeast	Southwest	Northwest
Alpha				
^{238}Pu (pCi/g)	39	89	90	174
$^{239,240}Pu$ (pCi/g)	18	34	65	42
^{241}Am (pCi/g)	<1.7	<1.7	<1.7	<1.7
^{237}Np (pCi/g)	5.1	2.5	2.0	3.2
Total alpha activity (pCi/g)	3,263	923	795	1,317
Beta				
^{99}Tc (pCi/g)	6,388	140	96	73
Total beta activity (pCi/g)	7,892	680	466	994
Gamma				
^{137}Cs (μCi/g)	<0.001	<0.001	<0.001	<0.001
^{95}Zr-Np (μCi/g)	<0.001	<0.001	<0.001	<0.001
^{106}Ru (μCi/g)	<0.001	<0.001	<0.001	<0.001
Total gamma activity (μCi/g)	<0.001	<0.001	<0.001	<0.001

[a]Activity analyses by radiochemical techniques.

Table B.4. Uranium isotopic analysis of the S-3 disposal pond sediment[a]

Nuclide	Assay (weight percent)			
	Southeast Pond	Northeast Pond	Southwest Pond	Northwest Pond
^{235}U	0.52	0.62	0.37	0.49
^{238}U	99.50	99.40	99.60	99.50

[a]Uranium assay determined by thermal emission ion counting mass spectrometry.

Appendix C

PRELIMINARY DRAFT OF WASTE ACCEPTANCE CRITERIA FOR THE CENTRAL WASTE DISPOSAL FACILITY*

C.1 INTRODUCTION

The criteria presented in this document, dated July 31, 1984, are to be used to define which wastes produced by the operations of the U.S. Department of Energy (DOE) facilities in Oak Ridge can be accepted for disposal at the Central Waste Disposal Facility (CWDF). The criteria are to be applied to the waste as it is received at the site, unless otherwise noted.

C.2 RADIOLOGICAL CRITERIA

1. Only solid waste that is contaminated or suspected of being contaminated with low levels of radioactivity shall be accepted at the CWDF.

2. The maximum radionuclide concentrations in the waste that shall be accepted for disposal at the CWDF are presented in Tables C.1 and C.2. The concentrations for radionuclides listed in Tables C.1 and C.2 are being revised as part of the pathways analysis effort.

   The concentrations are to be determined on the basis of the packaged waste as it is delivered to the CWDF.

3. The maximum disposal concentrations for radionuclides not listed in Tables C.2 and C.2 are to determined as a result of the pathways analysis.

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*Issued by CWDF project on July 31, 1984, and is to be finalized upon completion of pathways analysis.

Table C.1. Maximum short-lived radionuclide concentrations in the waste that shall be accepted for disposal at the Central Waste Disposal Facility (Ci/m^3 unless noted)

Short-lived radionuclide	Max permissible concentration
Total of all nuclides with half-lives less than 5 years	700
^{3}H	40
^{60}Co	700
^{63}Ni	3.5
^{63}Ni in activated metal	35
^{90}Sr	0.04
^{137}Cs	1.0

Table C.2. Maximum long-lived radionuclide concentrations in the waste that shall be accepted for disposal at the Central Waste Disposal Facility (Ci/m^3 unless noted)

Long-lived radionuclide	Max permissible concentration
^{14}C	0.8
^{14}C in activated metal	8.0
^{59}Ni in activated metal	22.0
^{94}Nb in activated metal	0.02
^{99}Tc	0.3
^{129}I	0.008
Uranium	0.05
Alpha-emitting transuranic nuclides with half-lives greater than 20 years	100.0 nCi/g
^{241}Pu	350 nCi/g
^{242}Cm	2000 nCi/g

4. To determine the maximum allowable disposal concentration for a mixture of radionuclides, the following sum of fractions rule applies. Each individual nuclide concentration in the waste divided by the maximum acceptable disposal concentration for that nuclide provides a number that is a fraction of 1.0. The sum of the fractions must be applied to the radionuclides listed in Tables C.1 and C.2 separately, and if the sum of the fractions of either table (not both tables) exceeds 1 then the waste is not acceptable. For example, if the sum of the fraction of Table C.1 is 0.9 and the sum of the fraction of Table C.2 is 0.6, the waste is acceptable.

5. The concentrations listed in Table C.1 apply to the wastes that are acceptable for disposal at the CWDF, but performance of the pathways analysis may indicate that the levels in Table C.1 would be unacceptable for disposal. The concentrations established by the pathways analysis would take precedence. However, since major factors used to determine the maximum disposal concentration in the waste were radionuclide leach rate and total amount leachable, it may be possible to stabilize the waste form (e.g., solidification in a cement matrix), thereby allowing for higher disposal concentrations. At no time shall the maximum allowable disposal concentrations exceed the levels listed in Table. C.1.*

6. The radiation reading at contact with the outer surface of the unshielded package shall not exceed the limits established in the ORNL Health Physics Procedure Manual, Procedures 4.1 and 5.1.

7. The exterior transferable surface contamination measured by standard smear method on the waste packaging shall not exceed the limits stated in Sect. C.4, Contamination Control, Criterion 4.

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*The method of testing the stabilized waste for leach rate and maximum amount of leachable radionuclides has not been established.

8. External shipment radiation dose rates/surface contamination limits for waste transport vehicles shall comply with 49 CFR 173.

C.3 PHYSICAL/CHEMICAL REQUIREMENTS

1. Liquid radioactive waste must be packaged with sufficient absorbent materials to absorb twice the volume of the liquid. This criterion must be met by the generator prior to shipment to the CWDF.

2. Solid/solidified radioactive waste shall contain less than 0.5% by volume of free liquid (drainable liquid from the waste form or freed standing liquid) when delivered to the CWDF.

3. No hazardous wastes (or co-contaminated) as defined by the Resource Conservation and Recovery Act or Tennessee Department of Public Health and Safety shall be accepted at the CWDF.*

4. Pathogenic and infectious material shall be autoclaved by the generator prior to shipment to the CWDF.

5. Pyrophoric wastes are not acceptable.

6. Waste shall not be readily capable of detonation or of explosive decomposition or reaction at normal pressures and temperatures or of explosive reaction with water.

7. Powders, ash, and similar respirable particulates shall be treated and stabilized or otherwise contained to reduce radiation or chemical exposure potential to personnel during staging, transport, handling, and disposal operations.

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*Asbestos- and beryllium-contaminated LLW may be accepted for disposal with the approval of the state. A separate disposal area may be required for these materials. All applicable Occupational Safety and Health Act and National Emission Standards for Hazardous Air Pollutants requirements shall be complied with.

8. Chelating agents that include amine polycarboxylic acids (e.g., EDTA, DTPA), hydroxy-carboxylic acids, and polycarboxylic acids (e.g., citric acid, carbolic acid, and glucinic acid) shall be kept to a minimum not to exceed 0.1% by weight of the waste (excluding package), unless it is proven by the generator that the presence of chelating agents in concentrations exceeding the above limit does not result in enhanced migration of radioactivity.

9. The waste shall not contain or be capable of generating quantities of toxic gases, vapors, or fumes harmful to persons handling, transporting, or disposing of wastes.

10. The waste shall not have a flash point below 140°F.

11. Solidified/stabilized wastes shall meet the following criteria.

    - o The waste shall be resistant to biodegradation.

    - o The waste form shall remain stable under compressive loads inherent in the disposal environment, unconfined compressive strengths of 50 psi or greater or exhibiting an unconfined penetration resistance of at least 50 psi (ASTM-C-403-70).

    - o The waste form shall remain stable if it is exposed to moisture (saturated or unsaturated) after disposal.

    - o The waste form shall be resistant to degradation caused by radiation effects.

    - o The waste shall meet any performance requirements as identified in the pathways analysis for solidified/stabilized waste and at no time exceed the radionuclide concentrations for waste acceptance listed in Tables C.1 and C.2.

12. The maximum size and weight for packaged waste that is acceptable for receipt at the CWDF will be established during the design of the facility.

C.4 WASTE PACKAGING REQUIREMENTS

1. General

Radioactive wastes transported to the CWDF shall be packaged in accordance with the Department of Transportation requirements contained in 49 CFR 173, "Shippers - General Requirments for Shipments and Packagings," to ensure that wastes can be safely handled, transported, and disposed.

2. Packaged Waste

- All containers of solid radioactive waste shall be labeled as containing radioactive material.

- Voids in the packages shall be kept to a practical minimum by either compacting the material or filling the voids with a material such as sand, gravel, or vermiculite is recommended.

- Only wastes packaged in weathertight containers shall be allowed for unloading at the disposal site during periods of precipitation. Cardboard boxes, nonwrapped bales of waste, and other package types that are not protected from weather shall not be accepted during periods of precipitation.

3. Unpackaged (Bulk) Waste

The requirements of 49 CFR 173.392, "Low Specific Activity Radioactive Material," apply to the shipment of bulk quantities of low specific activity radioactive materials.

4. Contamination Control

The removable radioactive contamination on waste packages when averaged over any area of 300 cm^2 of the package shall not exceed the levels stated as follows:

Contaminant	Maximum permissible level* (dpm/cm^2)
Natural or deleted uranium and natural thorium	
Beta-gamma	220
Alpha	22
All other beta-gamma-emitting radionuclides	22
All other alpha-emitting radionuclides	2.2

C.5 DOCUMENTATION

A waste disposal request form† properly completed by the generator must accompany the waste shipment to the disposal site. The following information shall be required:

- o weight of waste
- o volume of waste (m^3)
- o total activity present
- o radiation reading at container surface
- o radionuclides present
- o activity by radionuclide (Ci/m^3); for uranium (Ci/g)
- o physical description of waste
- o chemical description of waste

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*ORNL's current MPC standards are $\alpha \leq 30$ dpm/100 cm^2 and $\beta-\gamma \leq 1000$ dpm/100 cm^2. The values presented above represent 10% of the values from 49 CFR 173.397, "Contamination Control," as prescribed in 173.397, (a)(1).

†A standard form similar to UCN-2822, and in accordance with 10 CFR 61, will be developed for the purpose of documentation prior to operation of the CWDF.

- o special concerns or prescribed handling procedures, i.e., asbestos
- o solidification agent used (if applicable).

The generator, transporter, and disposer shall retain a record copy.

C.6 NONCOMPLIANCE

A generator submitting material to the CWDF in violation of these requirements (e.g., above allowable contamination limits, improperly packaged, hazardous, etc.) will be responsible for the costs of any corrective actions (exhumation, return, repackaging), including any studies/analyses required to determine corrective actions and the costs of any lost time, and will be subject to return of the material to assume the responsibility of the waste.

REFERENCES

Adams, J. A., and V. L. Rogers. 1978. A Classification System for Radioactive Waste Disposal -- What Waste Goes Where? NUREG-0456, U.S. Nuclear Regulatory Commission.

Auerbach, S. I. 1971. "Ecological Considerations in Siting Nuclear Plants. The Long-Term Biota Effects Problem." Nucl. Saf. 12(1): 25-34.

AEC (U.S. Atomic Energy Commission). 1974. Proposed Final Environmental Statement, Liquid Metal Fast Breeder Reactor Program, Vol. II, Environmental Impact of the LMFBR. WASH-1535.

Blaylock, B. G., and J. P. Witherspoon. 1976. "Radiation Doses and Effects for Aquatic Biota Exposed to Radioactive Releases from LWR Fuel-Cycle Facilities." Nucl. Saf. 17(3):351-361.

Boyle, J. W., et al. 1982. Environmental Analysis of the Operation of Oak Ridge National Laboratory (X-10 Site). ORNL-5870, Oak Ridge National Laboratory.

Code of Federal Regulations (40 CFR 190). 1977. Title 40: Protection of the Environment; Part 190: "Environmental Radiation Protection Standards for Nuclear Power Operations."

DOE (U.S. Department of Energy). 1983a. "Chapter III -- Management of Low-Level Waste." DOE Order 5820, draft dated December 16, 1983.

DOE (U.S. Department of Energy). 1983b. Radiological Guidelines For Application to DOE's Formerly Utilized Sites Remedial Action Program. Chap. 3, ORO-831 (Rev.), U.S.-DOE Oak Ridge Operations.

DOE (U.S. Department of Energy). 1981. "Requirements for Radiation Protection." Chap. 11, DOE Order 5480.1 and DOE Order 5480.1A.

DOE (U.S. Department of Energy). 1984. Draft Environmental Impact Statement--Central Waste Disposal Facility for Low-Level Radioactive Waste, Oak Ridge Reservation, Oak Ridge, Tennessee. DOE/EIS-0110-D.

Dunning, D. E., Jr., G. G. Killough, S. R. Bernard, J. C. Pleasant, and P. J. Walsh. 1981. Estimates of Internal Dose Equivalent to 22 Target Organs for Radionuclides Occurring in Routine Releases from Nuclear Fuel-Cycle Facilities. ORNL/NUREG/TM-190/V3, Oak Ridge National Laboratory.

Early, T. O., D. R. Drewes, G. K. Jacobs, and R. C. Routson. 1982. Geochemical Controls on Radionuclides Releases from a Nuclear Waste Repository in Basalt: Estimated Solubilities for Selected Element. RHO-BW-ST-39-P, Rockwell Hanford Operations.

Ebasco Services, Inc. 1984. Conceptual Design Report for Central Waste Disposal Facility. Prepared for UCCND.

Felmy, A. R., D. C. Girvin, and E. A. Jenne. 1982. MINTEQ -- A Computer Program for Calculating Aqueous Geochemical Equilibria, Pacific Northwest Laboratory, prepared for USEPA.

Frigerio, N. A., K. F. Eckerman, and R. G. Howe. 1975. "Background Radiation as a Carcinogenic Hazard," Radiat. Res. 62:599.

Garner, R. J. 1971. "Transfer of Radioactive Materials from the Terrestrial Environment to Animals and Man." Crit. Rev. Environ. Control 2:337-385.

Garrels, R. M., and C. L. Christ. 1965. Solutions, Minerals, and Equilibria. Harper and Row.

Gilbert, T. L., et al. 1983. Pathways Analysis and Radiation Dose Estimates for Radioactive Residues at Formerly Utilized MED/AEC Sites. ORO-832 (Rev.), US-DOE Oak Ridge Operations.

Healey, J. W. 1977. An Examination of the Pathways from Soil to Man for Plutonium. UC-41, Los Alamos Scientific Laboratory.

Hill, G. S. 1979. Dose for Various Pathways to Man Based on Unit Concentrations of Radionuclides Pertinent to Decontamination and Decommissioning of Properties. ORNL/OEPA-7, Oak Ridge National Laboratory.

Killough, G. G., and L. R. McKay, eds. 1976. A Methodology for Calculating Radiation Doses from Radioactivity Released to the Environment. ORNL-4992, Oak Ridge National Laboratory.

Ketelle, R. H., and D. D. Huff. 1984. Site Characterization of the West Chestnut Ridge Site. ORNL/TM-9229, Oak Ridge National Laboratory.

Kocher, D. C. 1981. Dose-Rate Conversion Factors for External Exposure to Photons and Electrons. ORNL/NUREG-79, Oak Ridge National Laboratory.

Miller, C. W., et al. 1980. Recommendations Concerning Models and Parameters Best Suited to Breeder Reactor Environmental Radiological Assessments. ORNL-5529, Oak Ridge National Laboratory.

NRC (U.S. Nuclear Regulatory Commission). 1976. Supplement to the Final Environmental Statement, Barnwell Nuclear Fuel Plant. NUREG-0082.

NRC (U.S. Nuclear Regulatory Commission). 1977. Calculation of Annual Doses to Man from Routine Releases of Reactor Effluents for the Purpose of Evaluating Compliance with 10 CFR Part 50, Appendix I, Revision 1. Regulatory Guide 1.109, Office of Standards Development.

NRC (U.S. Nuclear Regulatory Commission). 1981. "Proposed Rule 10 CFR Part 61, Licensing Requirements for Land Disposal of Radioactive Waste." Fed. Regist. 46(142):38081.

NRC (U.S. Nuclear Regulatory Commission). 1982. Final Environmental Impact Statement on 10 CFR Part 61, "Licensing Requirements for Land Disposal of Radioactive Waste". Vol. 1, NUREG-0945, Office of Nuclear Safety and Safeguards.

Parkhurst, D. L., D. C. Thorstenson, and L. N. Plummer. 1980. "PHREEQE-A Computer Program for Geochemical Calculations." Water Resources Investigations 80-96, U.S. Geological Survey.

Seeley, F. G., and A. D. Kelmers. 1984. Geochemical Information for the West Chestnut Ridge Central Waste Disposal Facility for Low-Level Radioactive Waste. ORNL-6061, Oak Ridge National Laboratory.

Woodward-Clyde Consultants. 1984. Subsurface Characterization and Geohydrologic Site Evaluation - West Chestnut Ridge Site. Vol. II, ORNL/Sub/83-64764/1&V2.

Yeh, G. T., and D. S. Ward. 1980. FEMWATER: A Finite-Element Model of WATER Flow Through Saturated-Unsaturated Porous Media. ORNL-5567, Oak Ridge National Laboratory.

Yeh, G. T., and D. S. Ward. 1981. FEMWASTE: A Finite-Element Model of WASTE Transport Through Saturated-Unsaturated Porous Media. ORNL-5601, Oak Ridge National Laboratory.

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